PILOT ARTIFICIAL RECHARGE SCHEMES: TESTING SUSTAINABLE WATER RESOURCE DEVELOPMENT IN FRACTURED AQUIFERS

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PILOT ARTIFICIAL RECHARGE SCHEMES:

TESTING SUSTAINABLE WATER RESOURCE DEVELOPMENT IN FRACTURED AQUIFERS

Final report to the

WATER RESEARCH COMMISSION

by

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EXECUTIVE SUMMARY

Full scale artificial recharge schemes for augmenting primary aquifers are in operation at Atlantis in the Western Cape and in the Omaruru Delta in Namibia confirming the viability of the technique for such aquifers. An initial assessment and feasibility study of artificial recharge schemes in Southern Africa identified the need for testing the concept in secondary aquifers, and in particular, fractured, hard-rock environments. Artificial recharge to South African aquifers is not a new concept as farmers in many parts of Southern Africa have constructed earth dams for increasing the recharge and thus augmenting their borehole supplies. However, generally no scientific information is available on the effectiveness of such artificial recharge schemes.

RESEARCH AIMS AND PRODUCTS

The following ***research aims*** were set out in the project proposal:

1. To test the artificial recharge concept in South African secondary aquifers.
2. To demonstrate the potential for artificial recharge in South African secondary aquifer to hydrogeologists and water resource planners.
3. To train local water resource managers in the operation and management of the pilot artificial recharge schemes.

The corresponding ***research products*** were as follows:

	Research Products	*Target Groups*
1.	One or more fully operational artificial recharge sites.	Rural communities or municipalities.
2.	Artificial recharge operation and management guidelines [developed from the pilot site(s)].	Artificial recharge scheme managers.
3.	Report on experience gained from the test site(s).	Hydrogeologists, Water Resource Planners.

PILOT ARTIFICIAL RECHARGE STUDY SITE SELECTION

As set out in the aims, the concept of artificial recharge of secondary aquifers had to be tested and for this purpose study sites had to be identified, developed and evaluated. For this purpose a number of hydrogeologists in various organizations, including the Department of Water Affairs and Forestry, were approach for suggesting candidate study sites. Initially ten sites were identified for follow-up, and subsequently six of these were selected for further evaluation. Key factors which determined the final selection of four sites were:

- The availability of water of suitable quantity, quality and reliability for recharging the aquifer
- The hydrogeological suitability of the area and the hydraulic characteristics of the particular secondary aquifer
- The probability of obtaining good data which would enable the quantification of artificial recharge, over and above natural recharge
- The potential for clogging of recharge basins, trenches or boreholes
- The possibilities of groundwater recovery
- The availability of funds for the construction of the scheme. This cost was not being covered by the WRC
- Management requirements, including the readiness of the local authority to start constructing the scheme.

The four study sites represented widely divergent situations. Three of the sites, Windhoek, Calvinia, and a small Namaqualand village, Karkams, are underlain by secondary aquifers and high quality water was needed for borehole injection. At the fourth site, Polokwane (formerly named Pietersburg), alluvium overlies fractured gneisses allowing the infiltration of treated municipal wastewater from the river bed into the fractured gneiss. The scale of water supply varied from a city (Windhoek) requiring in excess of fifteen million m^3/a to a community water supply (Karkams) needing only a few thousand m^3/a. This wide range of situations was considered important for demonstrating the applicability of the technique.

WINDHOEK PILOT ARTIFICIAL RECHARGE STUDY

Windhoek is located in the semiarid central highlands of Namibia close to water divides separating catchments stretching in all directions. Therefore, only limited quantities of surface water can be collected in the immediate vicinity of the city and the main supplies must be imported over long distances.

The geology of the Windhoek study area is extremely complex as a result of the geological processes associated with intra continental rifting and continental convergence. The Windhoek aquifer consists of interbedded quartzites and schists extending northwards from the Auas Mountain range. These east-west striking metasediments dip 15° - 30° to the north and are overlain by sandy colluvium filling a graben structure in the central, northern part of the study area. The hydrogeology of the area is dominated by the quartzite and schist horizons, and the faults and fractures that are prevalent throughout the Windhoek aquifer. The quartzites, being brittle, are highly fractured as a result of folding and faulting and have developed secondary porosity and permeability. The schists on the other hand are ductile and do not have well-developed secondary permeability. Highly productive boreholes are located along faults in the quartzites and the Windhoek aquifer serves as an important local water source.

The abstraction of approximately 103 Mm^3 water from the aquifer over the past 51 years has caused a gradual decline in groundwater levels which means that the average abstraction rate of 2.1 Mm^3/a exceeds the natural recharge rate. The additional water was considered to have been drawn from storage in the quartzites, mainly in the Auas Mountains. At the same time, the decrease in water levels created ample storage space for artificial groundwater recharge.

Between July 1998 and September 1999 three borehole injection tests were done on the Group 9 and 12 boreholes under the supervision of the CSIR. In these tests, potable water from the Windhoek water supply system was injected via deep boreholes into permeable parts of the aquifer. The longest test lasted 195 days and 289 000 m^3 of water were injected in borehole 9/8A. In this case the water was filtered through activated carbon and chlorinated before injection. The high injection rates of up to 59.4 L/s achieved during these tests were very encouraging and confirmed the feasibility of a full scale injection scheme.

Conclusions at Windhoek

The injection of water had a measurable effect on the water levels in the Windhoek aquifer. This was confirmed by a significant water level rise of several metres, up to 1.3 km from the injection boreholes. The injection rates were such that construction of a full scale artificial recharge scheme can be considered for the city. Assuming that borehole clogging during injection is either not a problem, or that it can be adequately managed, the rate at which water can be injected into the three tested boreholes is 1.65 Mm^3/a.

Further phases were foreseen which would include the drilling of special injection boreholes and the creation of the necessary infrastructure which would allow artificial recharge also in other parts of the aquifer.

It was concluded that the subsurface storage of surplus surface water would provide a sustainable groundwater supply also for drought periods and reduce evaporation losses from surface impoundments. Furthermore, it would also allow the natural recharge to gradually replenish the quartzites in the Auas

Mountains.

The quality of the injection water was good and the transient changes in water quality observed during recharge could be explained. Provided granular activated carbon filtration and subsequent chlorination are implemented, the injected water should not have any negative impact on the Windhoek aquifer.

Recommendations to Windhoek

Following the success of the pilot scale recharge tests, full scale implementation of borehole injection was recommended. A phased approach was proposed with the initial phase largely using existing infrastructure and production boreholes. This would also concentrate on filling the main cone of depression in the southern wellfields. Later phases would include special injection boreholes and additional infrastructure for injecting water closer to the mountains and utilizing the vast storage in that area.

It was also recommended that poor quality water should be prevented from entering the aquifer and the water should be fed through a granular activated carbon filter and be chlorinated prior to injection. Chemical deposits and silt in the reservoirs and injection pipelines should be removed prior to injection runs.

CALVINIA PILOT STUDY

Calvinia is situated in an arid area near the western edge of the escarpment. It receives winter rain but experiences periodic droughts and is often fully dependent on groundwater for its water supply.

About 12 km east of Calvinia is a roughly cylindrical brecciated pipe that contains open fractures and cavities which could be used for sub-surface water storage. The Kopoasfontein breccia pipe is intensely fractured and brecciated to a depth of 250 m. Its usable storage depends on the depth of the column of the pipe that is utilised, and geophysical evidence suggests that porosity is highest up to 182 mbgl. From abstraction and geophysical data, the volume of water available up to a depth of 182 m is estimated to be 80 000 m^3.

The brecciated pipe was filled with alkaline water with fluoride and other nuisance constituents and this water had to be removed before recharge with good quality surface water could commence. Several hydrochemical tests were carried out in an attempt to predict what quality water would be abstracted from the pipe after injection.

Conclusions at Calvinia

With appropriate management and the right size pump, Calvinia could store approximately 80 000 m^3 of water reserves in the breccia pipe as a sub-surface reservoir for emergency needs. These reserves are equivalent to at least two to three months supply.

The existing pump only allows partial use of the storage as it can only draw the water level down to 142 m. This is roughly equivalent to the storage of one month's water supply. Drawing the water level down another 40 m, could add at least another month's back-up to the water bank.

The ambient water quality in this geologic structure is unsuitable for human consumption, mainly due to fluoride and arsenic concentrations. The water can only be used for domestic consumption over short periods when diluted with good quality surface water.

Treated dam water is the most suitable recharge source. The injected dam water will be affected by the unusual geochemistry of the breccia pipe system and the hydrochemical conditions will need to be monitored on a regular basis until the pattern of water quality changes is established.

The safe residence time of the stored water will depend on the process of blending with deeper groundwater within the pipe, on water-rock interactions and on the effect of introducing oxygen into an anaerobic system. With repeated injection runs, it is hoped that the safe residence time of stored water in the pipe will increase. Whether or not it will eventually be possible to use this water without any blending with dam water prior to consumption, is unknown at this stage.

Recommendations to Calvinia

It is recommended that the abstracted pipe water be pumped into the surface impoundment prior to introducing this water into the reticulation system. This is in preference to pumping it directly to the mixing chamber before the reservoir. The reason for recommending this is because the concentrations of the elements of concern may decrease due to: i) the oxidising conditions in the dam; ii) the modified chemical equilibria after dilution with the dam water; and iii) adsorption on clay particles.

In order to take full advantage of this sub-surface reservoir, it is recommended that a suitable pump for dewatering the breccia pipe to a depth of 180 m be obtained. The breccia pipe should be artificially recharged each time after it has been drained. The water used for injection should be very low turbidity, fully treated dam water.

The quality of the water abstracted from the pipe for consumptive use should be monitored on a regular basis. A geochemist must assess these results, and advise on whether the water is suitable for consumption, or what dilution ratios with dam water are required.

KARKAMS FULL SCALE INJECTION SCHEME

The village of Karkams has a mean annual rainfall of 250 mm, and a population of 1690, depends solely on groundwater. The town is supported by three boreholes, of which G39002, the borehole used in the injection tests, is the lowest yielding. This borehole was chosen for the pilot artificial recharge study because it had previously been used for injection. Artificial recharge was initially recommended at this site by DWAF.

The Karkams aquifer consists of porphyritic granites, on which borehole G39002 is located, and older, fine-grained granitic gneisses. Most of the high yielding boreholes have been sited on the north-south striking faults that cut through this mountainous area. The groundwater salinity in the region generally increases from east to west and is directly related to elevation above sea level, which in turn, controls precipitation. The fluoride content of the groundwater in the area is generally high as is often the case in granitic and gneissic terrains. The trend generally follows that of salinity, with the highest values found in the low lying areas in the west. The groundwater quality at the artificial recharge site deteriorates with depth. It appears that at a depth of about 60 m to 70 m the water quality changes from relatively young or fresh water to older, more stagnant water.

Two controlled injection tests were conducted with the upgraded infrastructure using sand-filtered water from the adjacent ephemeral river. Following earlier injection runs with low salinity river water in 1995 and 1996 it was found that the abstracted water gradually became saline again. Prior to controlled injection in 1999 the EC value rose gently until it reached 292 mS/m. The test was conducted during the 28 days of streamflow in 1999, when 1 042 m^3 of water was injected into the abstraction borehole G39002. The second test took place in the year 2000, and the test ran for 90 days with 4 162 m^3 of water being injected into the newly constructed injection borehole G45757.

Conclusions at Karkams

The tests demonstrated that filtered river water can be injected into boreholes in the granitic gneiss at a rate of 0.5 to 0.7 L/s over a period of at least 90 days. The highest injection rate achieved over a shorter period was 1.4 L/s into production borehole G39002. The efficiency of the sand filter affected the injection rate.

The ephemeral river supplying the injection water flowed for one to six months each year from 1995 to 2000, except in 1998 when there was no flow. This implies that from 1 300 m^3 to 7 800 m^3 of water can be injected depending on the rainfall and river flow. This amounts to 0.4 to 2.2 times the borehole's sustainable yield.

An additional benefit of introducing this fresh water to the aquifer, is that it improves the groundwater quality. The more water that is injected the more the salinity decreases and the longer it remains at lower levels. However, the fluoride level remains above the recommended levels.

Recommendations to Karkams

The continued success of this scheme will depend on whether the management recommendations as set out in the recently drafted operating manual are adhered to. This is a very simple scheme to run, however, if the basic tasks of maintaining the sand filter are not done, then the scheme will lose efficiency and far less water will be available for recharge. Besides on-going monitoring of the system, it is recommended that support to the operators of the scheme be continued.

POLOKWANE (*FORMERLY PIETERSBURG*)ARTIFICIAL RECHARGE SYSTEM

Polokwane and Seshego, in the Northern Province, are largely dependent on surface water resources. However, they also have an elaborate groundwater abstraction infrastructure which can supply groundwater for domestic use in times of surface water shortages and during periods of peak demand.

Most of the study area is covered by alluvium, which extends to about 300 m either side of the of the Sand River drainage channel, reaching depths of up to 25 m. The deposits consist of upper clayey sands, overlying permeable coarser sands, and gravel boulder layers near or at its base. Underlying the alluvium deposits are the granite gneiss rocks of the Hout River Gneiss Complex. The granitic gneisses have probably been subjected to various phases of granitization. Pegmatitic and amphibolitic bands are common. Weathered basins and fractured gneisses form permeable zones to a depth of 60 meters.

The well fields are located along the banks of the Sand, Blood and Sterkloop Rivers. The present Sand and Blood River groundwater abstraction system predominantly utilise water from the deeper, secondary aquifer. Indications are that a proportion of the water abstracted from the Sand and Blood River well fields is derived from the infiltration of river water and the discharge of treated wastewater. This would mean that the treated wastewater is artificially recharging the aquifers supplying the town, which would imply indirect recycling of the treated wastewater. For this reason, concern exists over the longer term impact that waste water recharge could have on the quality of water in these aquifers.

Conclusions at Polokwane

The quality of water abstracted from boreholes in the gneisses located above the discharge point of the treated wastewater is generally good. This water differs in quality from the groundwater abstracted downstream of the discharge point. A close similarity in quality exists between water abstracted from the alluvium aquifer and the gneiss aquifer below the wastewater discharge point. This supports a potential hydraulic connection between the alluvium aquifer and the gneiss aquifer, and, therefore, the treated wastewater recharges the gneiss aquifer.

The major ion chemical data indicate that the treated wastewater does not pose serious risks to consumers. Slightly elevated nitrate concentrations during certain periods of the year could pose a health threat to infants. However, blending of the groundwater with the town's surface water sources can reduce the nitrate concentration and the salinity to acceptable levels.

Low levels of organic compounds are expected to be present in the treated wastewater and may lead to the formation of halogenated compounds when the water is treated with chlorine. Should such compounds be present in the domestic water supply they would represent a health threat to consumers.

Additional work could be done in order to optimise the functioning of the recharge system. The construction of recharge basins for intermittent use will allow filtration of the treated wastewater through an unsaturated zone. Such an approach would assist in the purification of the recharge water and will address concerns about the pollution of the groundwater by contaminants. In the present system it is possible that some bacteria, viruses or parasites could survive the bank infiltration process and contaminate the groundwater.

Recommendations to Polokwane

It is recommended that the Polokwane municipality implement the following as a means to optimise the present groundwater recharge process:

Implement a groundwater level monitoring programme, which include boreholes adjacent and away from the river. Ideally this should be done before and after the rainy season.

Assess the value of increasing infiltration of natural river flow by creating space within the aquifer during summer months. This could be done by abstracting groundwater at high rates during this period. Provided that the natural rainfall runoff is of better quality this could lead to improving the quality of the gneiss water.

Introduce a regular groundwater quality monitoring programme, which include constituents such as potassium and organic compounds, e.g. DOC. The treated wastewater discharge should be analysed for constituents such as phosphorus, total nitrogen, ammoniacal nitrogen, and nitrate nitrogen. If it is found that any of these compounds are of concern, then further pre-treatment of the waste water should be considered.

Depending on the presence of organic compounds in the abstracted water the final disinfected water should be tested for the presence of halogenated compounds.

Pollution sources that could impact the water quality in the well fields should be strictly managed. Sources of nitrate pollution, e.g. cattle feedlots could be of concern.

LESSONS LEARNT

The main factors determining the successful application of artificial recharge are:

- The quantity, type and reliability of the water source;

- The quality of recharge water, geochemistry and clogging issues;
- The hydraulic characteristics of the aquifer and groundwater recovery;
- Economics;
- Management requirements.

Recharge water can be obtained mostly from surplus surface water which is currently lost by means of evaporation from rivers and dams or which flows into the sea. Water for aquifer recharge purposes has to have a consistent quality and a fairly predictable quantity over time.

In contrast to water obtained from surface water impoundments in undisturbed catchments, river runoff, urban stormwater runoff and municipal wastewater, all have characteristics which complicate management and often necessitate some form of treatment before being usable for artificial recharge. This is exemplified by the variability of runoff periods and quantities at Karkams and Calvinia, or the quality of the treated wastewater in the Polokwane example.

Various factors influence the quality of the water when artificially recharging an aquifer. Infiltrating water moving through the unsaturated zone below a spreading basin, may undergo substantial quality changes before reaching the aquifer. The design of the system should be such that these changes will improve the quality of the water before it reaches the saturated zone. However, the initial quality of the recharge water often remains the main characteristic determining the final water quality, e.g. as at Polokwane.

The quality of the recharge water is a key variable in the decision on the type of artificial recharge system to be employed. High quality, low turbidity water can be utilised successfully in any kind of recharge system. However, when the water quality needs improvement, for example, to reduce the nutrient or organic compound, only a surface infiltration system involving soil aquifer treatment may be appropriate. Aquifer storage and recovery systems sometimes allow for a controlled degeneration in aquifer water quality where the recovered water is to be used for restricted purposes. In such cases the turbidity must still be low, but the chemical and/or microbiological quality may be impaired.

Geochemical modelling calculations proved to be a useful technique for predicting hydrochemical interactions in the case of Calvinia. It indicated that the breccia pipe needed to be dewatered of native groundwater before recharge in order to produce a potable end product.

The water used in borehole injection schemes has to be of a very high quality. Quality criteria include very low suspended solids, low concentrations of dissolved organic carbon, the absence of micro-organisms, and acceptable chemical compatibility between the injected water and the natural groundwater. The Windhoek case study is a good example.

The chemistry of the subsurface environment receiving the recharge water is also of concern, particularly in cases where the natural quality of the *in situ* groundwater is not of potable standard. An understanding of the rock types in the aquifer and the local groundwater chemistry is often sufficient to anticipate whether reactive minerals are present in the rocks. In some cases, e.g. in Windhoek, the composition of the recharge water is compatible with that of the in situ groundwater and problems are not anticipated. In contrast, at the Calvinia breccia pipe, the potential for reaction with the rocks after injection was a real concern and laboratory testing was applied for a more comprehensive geochemical study of the project feasibility.

Clogging of recharge basins, trenches or boreholes results in a reduction in the efficiency of the recharge process. Correctly dealing with the phenomenon of clogging, plays a decisive role in determining the success or failure of a scheme. Geochemical modelling can be a used to anticipate potential mineral precipitation reactions that could cause clogging.

Physical characteristics determining the suitability of an aquifer for accepting artificially recharged water are the aquifer's permeability and storage capacity. A third important factor is the aquifer's hydraulic gradient, which relates mostly to the recovery of the recharged water.

Aquifers which are highly permeable and which have high storage capacities are more suitable for receiving additional recharge water than those which have low permeabilities and low storage capacities. In relation to aquifer storage, the main concern with Southern African aquifers, and in particular, secondary aquifers, is whether there is sufficient storage space to accept the artificially recharged water. Where aquifers have been over-pumped, space will be available for artificial recharge. However, it should not only be considered as a means of replenishing over-abstracted aquifers, artificial recharge should be considered proactively as a means of maximising the use of available aquifer storage.

The recovery of artificially recharged water is seldom a problem in a well-utilised aquifer. In the case of Windhoek losses of recharged water from the aquifer are, in general, regarded as being insignificant. Even though losses of injected water are likely to be minimal, it was still necessary to develop an understanding of the aquifer's discharge area. The Karkams site has similarities to the Windhoek site, in that it may be necessary to induce flow towards abstraction boreholes by pumping either the injection boreholes or boreholes down gradient of the injection site. In Calvinia, the breccia pipe is hydraulically, virtually separate from the surrounding Karoo Sequence aquifer. This was evident after the constant discharge test.

Unused aquifer storage capacity can be developed at a significantly lower cost than surface storage facilities, and without the adverse environmental consequences frequently associated with surface storage. Often the overall costs of artificial recharge operations are less than half the capital cost of conventional water supply alternatives, especially those involving development of new reservoirs, treatment facilities or extensive pipelines. All costs should be considered also the savings from minimising water losses. A detailed economic feasibility study is being undertaken in Windhoek in order to establish the scale of the Windhoek artificial recharge facility.

The value of having a dependable quantity of water is usually considered to be of prime importance. In the case of Windhoek, the security of having an additional 25 - 30 Mm^3 in the city's water bank within five years would be of great value. Likewise in Calvinia, having an additional three months' supply in their water bank is considered to be a vital supplement to the town's vulnerable surface water resources.

Management of artificial recharge schemes commonly involve surface or waste water capture, treatment, pumping, distribution and water quality monitoring. In order for these processes to be efficient, careful planning and management is needed. One of the key management functions is to minimise clogging. The potential for clogging is especially high in *borehole injection systems*. For this reason it is essential that the high quality of the recharge water be maintained.

Artificial recharge and South Africa's National Water Act

The management and protection of South Africa's water resources is governed by the National Water Act of 1998. The Resource Directed Measures (RDM) form the main groundwater protection mechanisms, along with Source Directed Controls, under Chapter 3 of the National Water Act, 1998. The RDM includes Classification, the Reserve and Resource Quality Objectives. The Reserve aims to protect the quantity and quality of water required for aquatic ecosystems and basic human needs. The effect of artificially recharging an aquifer, would either increase the reliability of a groundwater resource or improve its quality, or both, and thus the effect on basic human needs should be positive. Only in cases where untreated water is used for artificial recharge could it have a negative impact on basic human needs.

Artificial recharge schemes need to be registered with the Department of Water Affairs and Forestry, and a licence for operating such schemes needs to be obtained (Part 1, Chapter 4, The National Water Act of 1998). Artificial recharge is specifically mentioned in Part 5, Chapter 4 of the 1998 Water Act, which is on "Controlled Activities". Here it states that "intentional recharging of an aquifer with any waste or water containing waste" will be classified as a controlled activity, and thus will require authorisation in addition to the standard licence requirements.

CONCLUSIONS

The objectives of this assessment of artificial recharge to secondary aquifers, have been reached.

Objective 1

To test the artificial recharge concept in South African secondary aquifers

Suitable secondary aquifers and test sites were identified and the artificial recharge concept tested in different situations. In particular, the large-scale Windhoek pilot study and the very small scale Karkams application confirmed the wide range of situations where the technique can succeed. This is of importance to large towns with extensive infrastructure and skilled staff, and to rural communities with minimal infrastructure and human resources.

The associated research product of one or more operational artificial recharge site could be delivered as the three sites Karkams, Windhoek and Calvinia are fully operational. In addition to these sites, Polokwane has been operational since the 1960's, and there may be a number of other towns, like Polokwane, that effectively recharge aquifers through waste water discharge.

Objective 2

To demonstrate the potential for artificial recharge in South African secondary aquifers to hydrogeologists and water resource planners

The technical success of the Windhoek artificial recharge pilot studies is confirmed on a management level by the fact that the Windhoek Municipality has decided to continue with the full-scale implementation of artificial recharge. This is to be implemented in the recommended phased approach set out in Section 1 of this report. The next phase of implementation is being funded by the Windhoek Municipality and is outside the scope of this project.

The results of the injection tests in Windhoek and Karkams were presented at seminars to the Groundwater Division of the Geological Society of South Africa, at a number of national and international conferences and at South African universities (Appendix 1). In addition to these presentations, the Karkams and Windhoek sites have been visited by a number of students, hydrogeologists and water resource managers. The case studies cover the third research product which is to report on experience gained from the test sites.

Objective 3

To train local water resource managers in the operation and management of the pilot artificial recharge schemes

During the pilot injection tests in Windhoek, Calvinia and Karkams, local water resource managers and operators were trained in the on-going operation and maintenance of the schemes. The second research product which entailed developing artificial recharge operation and management guidelines [developed from the pilot site(s)] was met in the case of Karkams, where a complete operating manual was compiled (Section 3, Appendix 3.4). This was the only site where three full injection runs were achieved and where it was possible to develop specific operational guidelines. Generic lessons learnt from all pilot study areas and general operating principles are described in Chapter 2 of this report.

This study was undertaken at an opportune time - the Windhoek municipality had recently completed a brief trial borehole injection test, and was ready to do more pilot injection tests, and the Calvinia municipality had been looking for options to increase the reliability of their water supplies. Both municipalities welcomed the opportunity to participate in this pilot artificial recharge study.

Karkams now has an effective artificial recharge scheme - even though it is a small-scale operation with a maximum injection rate of 1 L/s. The Windhoek municipality is currently implementing a large-scale borehole injection scheme with a planned injection rate of 200 L/s. The Polokwane waste water infiltration scheme remains a successful system of waste water management and re-use but needs evaluation regarding long term water quality concerns. The only site where it is premature to conclude the success of the scheme is at Calvinia, where more abstraction-injection runs are required before the value of this scheme will be known.

The concept of applying artificial recharge technologies in Southern African secondary aquifers has been demonstrated. A number of groundwater scientists and engineers have visited the sites, and fourteen papers based on this study have been presented at local and international conferences. What remains is for this technology to be implemented wherever the conditions are suitable in this water-strapped region of Africa.

Internationally, the concept of artificial recharge in aquifer management and in integrated water resource management is well established. Previously South Africa has only applied this technology to a primary, unconsolidated sand aquifer, namely the Atlantis aquifer. The pilot studies of this project have demonstrated that this technology can successfully be transferred to Southern Africa's predominantly fractured, hard-rock aquifers.

RECOMMENDATIONS

General recommendations regarding the development and promotion of artificial recharge in South Africa stemming from this research project are the following:

- The successful application of artificial recharge to both primary and secondary aquifers has been demonstrated and consideration should be given to using this technology in the following circumstances:
  - In areas where groundwater is being abstracted over and above natural recharge;
  - In areas with surplus seasonal surface water where groundwater abstraction is in balance with natural recharge, but where more water is required throughout the year;
  - In areas where groundwater is saline or non-potable;
  - In areas where a buffer zone may be required between good and poor quality groundwater (such as in the mining industry, where the migration of contaminated groundwater needs to be controlled);
  - In primary aquifers suitable for waste water discharge - like the Polokwane case. In these circumstances it is essential to treat the waste water to acceptable standards prior to artificial recharge.

- Strategic, large aquifers should be identified, such as the Table Mountain Group aquifers and the dolomitic aquifers, where conjunctive use between surface water and groundwater could be optimised. By utilising large aquifers in this manner, vast quantities of surface water that would either be lost by evaporation or flowing into the sea, could be stored and used on an annual basis, or reserved for emergency supplies. Detailed research would be required to assess the suitability of the target aquifers. For example, in the dolomites, water compatibility and sinkhole formation would need to be studied.

- This technology should be introduced in tertiary educational institutions. Water resource planners, water engineers and hydrogeologists should be taught the value of this technology and the key factors that influence its success.

- The value of artificial recharge should be conveyed to a wide audience who are currently involved in water resource planning. The audience should include planners involved in all areas of water supply and aquifer management, that is, municipal, rural, agricultural, industrial and mining. Artificial recharge needs to be viewed as an integral part of sustainable groundwater management and should be considered as a potential option in all groundwater development plans.

- In order to convey the artificial recharge technology, lectures throughout the country and demonstration site visits should be carried out. Supporting notes or booklets on theory, planning and case studies should form part of this technology transfer package.
- The Polokwane study needs follow-up research in order to quantify more accurately the effect of waste water infiltration into the gneissic hard-rock aquifer. Important data is currently unavailable. This data should be obtained, assessed and integrated with current knowledge prior to presenting this infiltration scheme as a model for waste water discharge - aquifer recharge schemes.

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CHAPTER 1:

Introduction

Contents

1. INTRODUCTION

This project follows on from the assessment and feasibility study of artificial recharge schemes in Southern Africa undertaken by Murray and Tredoux (1998). Four pilot study sites were identified where the artificial recharge concept could be tested in secondary aquifers. Three of these sites, Windhoek, Calvinia and Karkams, a small Namaqualand village, use borehole injection technology and one site, Polokwane (previously called Pietersburg), uses infiltration of municipal waste water from a river bed.

This report consists of a brief introductory section which describes the process of selecting the pilot study sites (Chapter 1), and generic lessons learnt from the artificial recharge test sites (Chapter 2); and four sections which describe, in detail each pilot artificial recharge study.

1.1 An overview of artificial recharge in South and Southern Africa

Artificial recharge to South African aquifers is not a new concept. Scattered throughout the country are small earth dams which farmers have built to augment their borehole supplies. In all but one case, so it seems, the impact of these artificial recharge schemes on groundwater has not been established. The one study where records were kept, was in the Soutpansberg District of the Northern Province, where DWAF recorded borehole yields and water levels prior to and after the construction of the earth dams (De Villiers, 1971). Unfortunately the information obtained is insufficient to establish the true effect of the recharge dams. Ideally, the data required should include water levels and rainfall, spanning a few years prior to, and after the construction of the dams. In Namibia, sand storage dams were constructed in many rivers, both for augmenting borehole yields and for serving as a water supply source. Wipplinger (1953) carried out extensive research in this regard.

The one artificial recharge scheme where records exist which demonstrate the effectiveness of the scheme, is in the dune sands, primary aquifer, at the town of Atlantis in the South Western Cape. A number of valuable lessons have been learnt over the last twenty years from this municipal waste water, infiltration scheme (Tredoux *et al.*, 1999). In Namibia, the Omaruru Delta (Omdel) scheme was constructed for capturing and recharging runoff into the river delta primary aquifer in the Namib (Tordiffe, 1996).

Pilot artificial recharge studies have been done in the primary aquifers of the Cape Flats in Cape Town (Tredoux and Wright, 1996) and near Maun in Botswana (Eastend Investments, 1997). The only reported pilot study to be done in a secondary aquifer is the three week injection test that was done on the Windhoek quartzite aquifer in 1997 (Carr Barbour & Associates, 1977). This project is the first systematic study of artificial recharge in Southern African secondary aquifers, and in particular, artificial recharge by borehole injection in fractured, hard rock aquifers.

1.2 Research aims

The research aims set out in the project proposal were:

1. To test the artificial recharge concept in South African secondary aquifers.
2. To demonstrate the potential for artificial recharge in South African secondary aquifers to hydrogeologists and water resource planners.
3. To train local water resource managers in the operation and management of the pilot artificial recharge schemes.

The planned research products described in the project proposal were:

	Research Products	*Target Groups*
1.	One or more fully operational artificial recharge sites.	Rural communities or municipalities.
2.	Artificial recharge operation and management guidelines [developed from the pilot site(s)].	Artificial recharge scheme managers.
3.	Report on experience gained from the test site(s).	Hydrogeologists, Water Resource Planners.

1.3 The process of selecting pilot artificial recharge study sites

The process of identifying possible artificial recharge pilot sites began by contacting a number of hydrogeologists. They included staff from DWAF, research institutions, groundwater consulting companies and a mining company. Ten sites were identified for follow-up, and six were finally selected for further study. The potential artificial recharge sites are listed below.

Suggested artificial recharge sites that warranted further study:

- Kenhardt
- Calvinia
- Williston
- Karkams
- A farm near Rustenburg
- Windhoek

Suggested artificial recharge sites which were rejected after a brief assessment:
- Strydenburg
- Graaff Reinet
- Beaufort West
- Dysselsdorp

The six sites selected for further study are all associated with secondary aquifers, and they have potential for success in terms of the following factors:

- The availability of water for recharging the aquifer;
- The hydraulic characteristics of the aquifer;
- The quality of the recharge water;
- Clogging of recharge basins, trenches or boreholes;
- Groundwater recovery;
- Economics;
- Management requirements.

The four rejected sites were considered unsuitable for further study because either one or more of the above factors were not met, or it was thought that they would not be met within the time frame required to get the project started.

Table 1 presents a summary of each of the selected sites.

Table 1 Suggested artificial recharge pilot study sites

TOWN	*AQUIFER*	*AR[1] METHOD*	*WATER SOURCE*	*COMMENTS*
Kenhardt	Alluvium and weathered/ fractured gneiss	spreading	Rooibergdam which usually fills during winter and is often dry during summer	*This case study would test the use of dam water for AR in typical weathered and fractured gneisses overlain by alluvium.* Kenhardt depends on a wellfield for all their water requirements. The wellfield cannot yield the seasonal peak demand, and it will not be able to meet future demands. AR, which was recommended by DWAF (Nonner, 1979), is probably the cheapest way to solve Kendardt's water resource problems. The more expensive alternatives include: develop a known aquifer 18 km from town; develop a known, but saline aquifer within 2 km from town - desalination would be required; develop a known, deep-seated aquifer 15 km from town; and pipe water 75 km from the Orange River. An advantage of using Kendardt as a pilot AR site is that an established monitoring system is in place and historical water level and abstraction data exists. This information would be useful in assessing the impact of an AR scheme. The conceptual design of this scheme has been developed (Murray and Tredoux, 1998).
Calvinia	Breccia plug in Karoo Sequence	injection	groundwater (existing boreholes) & the Karee dam	*This study would test the use of a transmissive groundwater compartment for surplus water storage.* A new wellfield has recently been developed to augment the town's water supplies. With little additional expense, a large groundwater storage system could be developed. The aim would be to transfer groundwater from an aquifer which is annually recharged, and/or treated water from the Karee dam, when available, to a groundwater compartment which is not regularly recharged. This water would then be available for use in times of high demand. Boreholes and monitoring systems exist, and the conceptual design of this scheme has been developed (Murray and Tredoux, 1998).

Table 1 continues overleaf...

TOWN	***AQUIFER***	***AR^1 METHOD***	***WATER SOURCE***	***COMMENTS***
Williston	Horizontal fracture in Karoo Sequence	injection	groundwater (existing borehole)	*This study would test the concept of AR to a confined aquifer with a horizontal fracture pattern.* Williston will soon need more water to meet their domestic water needs. An aquifer adjacent to the currently used aquifer could serve as an ideal recharge source. The possible AR system is conceptually extremely simple and may not require pumping. Williston's alternative is to pump groundwater about 8 km, which is an expensive alternative to this proposed AR scheme. Monitoring has taken place for 14 years. The conceptual design of this scheme has been developed (Murray and Tredoux, 1998).
Windhoek	Fractured quartzites	injection	treated dam water	*The study would test injection of treated surface water into a highly fractured, secondary aquifer.* Windhoek depends on three sources of water, namely surface reservoirs built in ephemeral rivers, groundwater and water reclamation from domestic waste. During 1997 the evaporation from the surface reservoirs was 37 x 10^6 m^3, while the consumption was 15.7 x 10^6 m^3. The aim of artificial recharge to the Windhoek secondary aquifer is to maximise the safe storage of surface water, and to minimise the losses associated with evaporation. The first pilot artificial recharge study was carried out in 1996/7. An advantage of the Windhoek study is that data on groundwater levels and abstraction have been recorded for 50 years.
Rustenburg farm	Igneous rocks , base of the Bushveld Igneous Complex	injection	spring	*This study would establish the effect of injecting spring water into low yielding boreholes.* A farmer in the Rustenburg area gravitates clear spring water during the summer rainfall period into 4 or 5 boreholes. During the winter, the water is abstracted from the boreholes for irrigation purposes. The boreholes yield (and receive from the springs) on average about 1 l/s, and the farmer says that he can abstract the same amount of water that he injects. The scheme has been operational for 3 years. It would be interesting and of use to geohydrologists to quantify the effect of this scheme on aquifer recharge. This study would involve establishing a monitoring system so that the effectiveness of this scheme can be established.

Table1 continues overleaf

TOWN	*AQUIFER*	*AR[1] METHOD*	*WATER SOURCE*	*COMMENTS*
Karkams	Granite	injection	surface runoff	*This study would establish the effect of injecting filtered surface water into a low yielding borehole.* As part of a groundwater supply scheme for this Namaqualand village, DWAF (Esterhuyse, 1990) recommended that one of the boreholes be used for injection and supply. Water from a small stream is fed through a sand filter up-slope of the borehole and then gravitated into the borehole. The scheme has been monitored since it came into operation in 1995 (Toens and Partners, 1997). Two problems have emerged: Firstly, the sand filter was not constructed (or designed?) to maximise filtration and flow efficiency, and is therefore not working properly. The borehole is currently not being recharged because sand and organic matter pass through the filter. Secondly, the rate at which water enters the borehole during artificial recharge is too high. The borehole gave a blow yield of 7.2 l/s during drilling, and was test pumped for 50 hours at 3.2 l/s. Due to the poor recovery after the constant discharge test (a residual drawdown of 3 m), a production yield of 0.3 l/s for 20 hours per day was recommended. The ± 50 mm diameter pipeline from the sand filter allows water to enter the borehole at a rate greater than that which the borehole can accept, and as a result, the recharge water overflows from the borehole. From the monitored data, which includes abstraction, rainfall, water levels and electrical conductivity, it is difficult to establish to what extent the aquifer accepts this recharge water. An observation borehole would need to be drilled near the injection borehole to establish this. Because the water is gravitated at far too high a rate into the borehole (so that the water flows out the top of the borehole), and because there are no observation boreholes, it is difficult to establish the effect that this scheme is having on aquifer recharge. In order to study this site, it would be necessary to re-design the scheme so that the effect of artificial recharge on the aquifer can be established.

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[1] Artificial Recharge

Key factors which influenced the selection of the four final pilot study sites were:

1. The hydrogeological suitability of the area (bearing in mind that we targeted secondary aquifers only);
2. The availability of water to recharge the aquifer (in terms of quantity, quality and reliability);
3. The probability of obtaining good data (data which would enable us to quantify artificial recharge, over and above natural recharge)
4. The availability of funds for the construction of the scheme. This cost is not being covered by the WRC;
5. The readiness of the local authority to start constructing the scheme.

Kenhardt was ruled out because of points 4 and 5. **Williston** was discarded because the municipality, although seeming to be committed to this scheme, did not follow up on their component of the planning of the project. The **farm near Rustenburg** was not seen to be suitable for this study because of point number three above.

The criteria for going ahead with the pilot study sites were all met in the cases of **Windhoek Calvinia** and **Karkams**. Karkams was already operational, but it was poorly designed. Since Kenhardt was not an option, Carel Haupt of Water Systems Management suggested **Pietersburg** which has a similar geology to Kenhardt (alluvium overlying fractured gneisses). This option was accepted, even though an artificial recharge scheme would not be constructed; rather the aim would be to understand the recharge that takes place as a result of waste water being discharged into an ephemeral river. Figure 1 shows the location of the four sites that were selected for detailed study. Note that shortly before compiling this final report, the name Pietersburg was changed to **Polokwane**, and the province's name was to be changed from the Northern Province to the Limpopo Province. In this report, the name Polokwane has been used, however, in the review by Dr P Dillon (which was done in 2000), the name Pietersburg was used.

1.4 Artificial recharge and South Africa's National Water Act

The management and protection of South Africa's water resources is governed by the National Water Act of 1998. The Resource Directed Measures (RDM) form the main groundwater protection mechanisms, along with Source Directed Controls, under Chapter 3 of the National Water Act, 1998. The RDM includes Classification, the Reserve and Resource Quality Objectives.

The Reserve aims to protect the quantity and quality of water required for aquatic ecosystems and basic human needs. The effect of artificially recharging an aquifer, would either increase the reliability of a groundwater resource or improve its quality, or both, and thus the effect on basic human needs should be positive. Only in cases where untreated water is used for artificial recharge could it have a negative impact on basic human needs. It is stressed in Chapter 2 of this report that the quality of artificially recharged water should generally be of a high standard, and that pre- and post-treatment of this water must be carefully studied prior to implementing an artificial recharge scheme. Where surface water is used for artificial recharge, the effect on aquatic ecosystems needs to be assessed. This may, in some cases limit the water available for artificial recharge.

The Resource Quality Objectives will aim to protect the desired functionality of the aquifer. Injecting treated, potable water could negatively impact aquifer integrity (matrix dissolution) and hypogean ecology. These aspects should be investigated prior to artificial recharge.

The RDM classification system, which is currently being developed (Xu et al, 2001), uses reference (or pristine) conditions as a bench mark. It will need to be noted that, whilst artificial recharge may further remove the hydrogeological system from a pristine condition, it will improve the overall utility of the aquifer. Therefore a change from a Class A (pristine) to a Class C (seriously modified) due to artificial recharge, will be a positive modification.

Artificial recharge would be classified as a "Water Use" in The National Water Act of 1998. Artificial recharge schemes therefore need to be registered with the Department of Water Affairs and Forestry, and a licence for operating such schemes needs to be obtained (Part 1, Chapter 4, The National Water Act of 1998). Artificial recharge is specifically mentioned in Part 5, Chapter 4 of the 1998 Water Act, which is on "Controlled Activities". Here it states that "intentional recharging of an aquifer with any waste or water containing waste" will be classified as a controlled activity, and thus will require authorisation in addition to the standard licence requirements.

Figure 1 Location of artificial recharge study sites

CHAPTER 2:

Success Factors

Contents

2. FACTORS AFFECTING THE SUCCESS OF ARTIFICIAL RECHARGE SCHEMES AND LESSONS LEARNT FROM THE PILOT INJECTION TESTS

This chapter highlights the success factors that affect the potential for artificial recharge and it draws on the lessons that were learnt from the borehole injection tests at Windhoek, Calvinia and Karkams, and on the study of the Polokwane infiltration scheme. The focus is on the borehole injection schemes, and the material is adapted from Murray and Tredoux (1998).

The main factors which determine whether artificial recharge is likely to be successful are:

- The quantity, type and reliability of the water source;
- The quality of recharge water, geochemistry and clogging issues;
- The hydraulic characteristics of the aquifer and groundwater recovery;
- Economics;
- Management requirements.

2.1 The quantity, type and reliability of the recharge water sources

Recharge water can mostly be obtained from surplus surface water which is currently lost by means of evaporation from rivers and dams or which flows into the sea. Water for aquifer recharge purposes has to have a consistent quality and a fairly predictable quantity over time.

Possible sources are municipal wastewaters, storm runoff, rainfall harvesting, river flows, and water releases from dams. *Municipal wastewater* has a predictable quantity and quality, but it may not necessarily be suitable for artificial recharge purposes for unrestricted reuse. It invariably requires significant chemical treatment before being considered of sufficiently high quality for aquifer recharge. After treatment, artificial recharge by means of infiltration may improve the water quality to a suitably high standard. *Urban storm runoff* is usually highly variable in quality, but with the exception of industrial runoff, is on average of higher chemical quality than municipal wastewater and agricultural runoff.

Due to the highly variable flow rate, *storm runoff* is usually collected in an impoundment or basin, from which controlled release of water into recharge basins takes place, after settling of the bulk of the suspended solids. *Rivers* have a more consistent quantity of flow than storm runoff in their catchment areas, due partly to a base-flow contribution from groundwater/ interflow, and a wide range of rainfall-response times.

In many parts of Southern Africa, and in particular the dryer western areas where evaporation is high, it may be beneficial to store surface water underground. In order to assess this, both the evaporation losses and the economics associated with treating, transferring and recovering the artificially recharged water need to be determined.

Out of the four study sites, Karkams is the only site which depends on an ephemeral river for injection water. Although in some years there will not be any water available for injection, this scheme can still be effective because when water is available, it can be used to rapidly replenish the groundwater that would have been pumped from storage. Similar "opportunistic" artificial recharge schemes that capture water when available could be developed in numerous small catchments throughout South Africa and elsewhere. This technique can improve the sustainability and the quality of rural water supplies even though the source water quantities may be variable and even unpredictable as at Karkams.

2.2 Quality of recharge water, geochemistry and clogging issues

Various factors influence the quality of the water when artificially recharging an aquifer. Infiltrating water moving through the unsaturated zone below a spreading basin, may undergo substantial quality changes before reaching the aquifer. The design of the system should be such that these changes will improve the quality of the water before it reaches the saturated zone. However, the initial quality of the recharge water often remains the main characteristic determining the final water quality.

2.2.1 Quality of recharge water

Characterising the water quality parameters is an important part of the initial feasibility study for artificial recharge schemes. Samples of the recharge water should be collected and analysed for a range of physical and chemical parameters (Table 2), with repeat sampling in cases where these parameters are likely to vary over time.

Table 2 Recommended parameters for geochemical investigations of artificial recharge

Recharge Source	*In Situ Groundwater*	*Aquifer Rocks Or Sediments*
pH EC or TDS Turbidity, Suspended Solids Major ions: Na, K, Ca, Mg, Cl, SO_4 Alkalinity Minor species: NH_4, NO_3, F, Fe, Mn, DOC Microbiology: indicator organisms	pH EC or TDS Eh or Dissolved Oxygen Major ions: Na, K, Ca, Mg, Cl, SO_4 Alkalinity Minor species: NH_4, NO_3, F, As, Fe, Mn, DOC Microbiology: SO_4 - reducing or Fe bacteria present?	Bulk composition Mineralogy Particle size/reactive area Leachable salts

The quality of the recharge water is a key variable in the decision on the type of artificial recharge system to be employed. High quality, low turbidity water can be utilised successfully in any kind of recharge system. However, when the water quality needs improvement, for example, to reduce the nutrient or organic compound, a surface infiltration system involving soil aquifer treatment may be appropriate. Alternatively, pretreatment options may be exercised to reduce turbidity and improve quality for direct recharge.

Also, aquifer storage and recovery systems (where the same borehole is used for recharge and recovery), sometimes allow for a controlled degeneration in aquifer water quality where the recovered water is to be used for restricted purposes. In such cases the turbidity must still be low, but the chemical and/or microbiological quality may be impaired. In general, the extent of post treatment of water recovered from an artificial recharge operation, will mainly depend on the intended use.

Geochemical modelling calculations are a useful technique for predicting whether sufficient dilution can be achieved to increase the quantities of potable water from a poor quality aquifer or for testing what modifications could be made to the recharge water chemistry to avoid or minimise unwanted chemical reactions. Modelling calculations for Calvinia indicated that the breccia pipe needed to be dewatered of native groundwater before recharge, because a high mixing fraction of about 80% surface water was needed to produce a potable end product.

The water used in borehole injection schemes has to be of a very high quality. Important quality criteria for successful injection include very low suspended solids, low concentrations of dissolved organic carbon, the absence of micro-organisms, and acceptable chemical compatibility between the injected water and the natural groundwater.

The water used for injection in Windhoek was taken from the city's water supply. This fully treated drinking water comes mainly from the dams that supply Windhoek, but a small proportion is derived from groundwater that is pumped from the municipal boreholes. The microbiological quality of the injected water was good (a heterotrophic plate count of 2 per 1 mL at 22° C after 72 hours). The internationally recognised target range for drinking water is 0 - 100 counts per 1 mL (Department of Water Affairs and Forestry, 1996).

In the case of the long duration injection test on Windhoek's 9/8A borehole, the water was further treated by percolation through a granular activated carbon filter, followed by chlorination. This was done to ensure a very low organic carbon content and to further filter the water prior to injection; and it was felt necessary because any particulate matter that may have accumulated in the reservoirs needed to be prevented from entering the aquifer.

Percolation of the water through the carbon filter mechanically screened the water, removing all remaining suspended material, which ensured both a very low turbidity (<< 1 NTU) and a low organic carbon content of approximately 3 mg/L DOC. Chlorination of the injected water (to 0.3 mg/L free chlorine), after passing through the carbon filter, ensured disinfection.

In the case of Calvinia, the water used for injection was also the town's water supply, which is fully treated dam water. Special care needs to be taken after rainfall events, when the dam water is very turbid in order to prevent this water entering the aquifer. It is also critical that flocculated matter is effectively removed in the treatment process before injection.

In Karkams, the water used for injection is surface runoff from an ephemeral stream. Because the borehole used for injection is located close to the source of the stream and in a remote area, the main concern was to remove suspended solids prior to injection. An appropriately designed in-stream sand filter ensures that the injected water is of very low turbidity.

In Polokwane treated waste water is allowed to infiltrate to a secondary gneiss aquifer through a bank filtration process. The quality of the waste water discharge is maintained within the national effluent quality standards. None of the constituents measured were found to occur in concentrations that pose a serious threat to water users. During the rainy season natural flow in the Sand River dilutes the recharge water and consequently improves the quality.

2.2.2 Aquifer geochemistry

The chemistry of the subsurface environment receiving the recharge water is also of concern, particularly in cases where the natural quality of the *in situ* groundwater is not of potable standard. In most cases, an understanding of the rock types in the aquifer and the local groundwater chemistry is sufficient to anticipate the presence of reactive minerals in the rocks, without having to submit rock samples for analysis (Table 2). More detailed investigation of the rock composition and the chemistry and variability of the local groundwater may be needed for modelling purposes, if problems are predicted.

In some cases, the composition of the recharge water will be compatible with that of the *in situ* groundwater and problems are not anticipated. This is especially true if the aquifer to be recharged comprises inert materials such as quartzite rock or clean quartzitic sands. If both water sources have a low content of dissolved solids, no detectable species of health concern (e.g. fluoride, arsenic or nitrate) and neutral pH and oxidising status, no further geochemical investigations may be warranted. At Windhoek, for example, the surface water, after treatment and filtration through a carbon column, should not cause any adverse chemical reactions in the aquifer.

In contrast, at the Calvinia breccia pipe, the potential for a reaction between the recharge water and the rocks after injection was a real concern and laboratory testing was applied for a more comprehensive geochemical study of the project feasibility. Various water rock interaction tests can be performed in the laboratory to add value to an artificial recharge investigation, including leaching tests, batch tests (shake tests) and column tests.

Leaching tests involve allowing the solid material to react with distilled water or solutions of acid or base and analysing the dissolved constituents in the final solution. Leaching of samples from the Calvinia breccia pipe showed that the rock contained large quantities of sodium which could be released if surface water is injected. In addition, the leachate also had a high alkalinity. The rocks also have the potential to release significant potassium and sulphate as well as some fluoride and ammonium into solution.

Batch tests are used to react known quantities of solid material with a solution of known composition. These tests can be used to quantify ion exchange or sorption processes that may occur during artificial recharge, particularly when there is a high proportion of clay minerals or organic matter in the aquifer.

Column tests aim to give a better approximation of natural conditions than batch tests. The recharge water is infiltrated through the column and the final solution is analysed as a function of time, giving breakthrough curves for various chemical species that may be leached or adsorbed during reaction with the rock. The solid material can also be investigated for precipitated solids or biological fouling. Column tests, however, are usually time-consuming and difficult to set up and so were not conducted for the case studies in this project.

A laboratory experiment can never entirely simulate the controlling parameters of the field situation over the same time scales of an operational recharge scheme. Laboratory tests also cannot replace pilot field experiments. The tests can, however, give valuable information about the likely success of a scheme and different management options.

2.2.3 Clogging issues

Clogging of recharge basins, trenches or boreholes results in a reduction in the efficiency of the recharge process. Correctly dealing with the phenomenon of clogging, plays a decisive role in determining the success or failure of a scheme. Clogging of the system can be due to mechanical, physical, chemical and biological processes, as well as a combination of these. It can take place at the infiltration surface, in the unsaturated zone, or in the aquifer itself. In the case of injection, it could block the fractures leading away from the borehole. A thorough understanding of the processes involved and the consequent reversibility or irreversibility of the situation is needed in order to be able to manage the clogging phenomenon.

The processes that are primarily responsible for clogging are: deposition of suspended solids from the recharge water; air entrapment and gas binding; biological growth of bacteria on or within the infiltration media and the surrounding formation; and chemical reactions between recharge water, groundwater and the aquifer material.

Geochemical modelling can be a used to anticipate potential mineral precipitation reactions that could cause clogging. Precipitation of carbonate minerals and iron oxyhydroxides, in particular, may clog up the aquifer and infrastructure if recharge activities cause large quantities of these solids to precipitate. Modelling calculations are particularly advisable where the groundwater has a high concentration of dissolved iron and the recharge water has high dissolved oxygen or where waters with high calcium and alkalinity will come in contact with a high pH environment.

Freely available modelling software packages such as PHREEQC (U.S. Geological Survey) or MINTEQA2 (U.S. Environmental Protection Agency) are suitable for performing this type of calculation. Input data needed are the chemical composition of the recharge and *in situ* groundwater and the nature of the subsurface environment (e.g. temperature, redox potential) as well as an estimate of the proportion of each water type in the mixture. By calculating the equilibrium water composition and mineral precipitation potential for a range of mixing fractions (from one end member water type to the other), the calculations can be used to simulate the effects across a mixing front when recharge water comes into contact with the water already in the aquifer.

In the Calvinia study PHREEQC software was used to establish whether calcium carbonate clogging may occur if Karoo groundwater were to be used as an injection source. This led to the rejection of Karoo recharge water in favour of surface water from the Karee Dam.

At Windhoek, because the injection water is lower in dissolved solids (TDS) than the groundwater, particularly with respect to calcium, the injection water dilutes the natural groundwater and, therefore, shifts the calcium carbonate equilibrium, reducing the precipitation potential.

On-going groundwater quality monitoring is required for the continuation of these pilot artificial recharge schemes. Some techniques for monitoring include sampling and analysis of the recovered water over time or logging of physical and chemical parameters with depth down a borehole using a multiprobe and datalogger. Long-term data collected from the monitoring will reveal the extent to which, and the rate at which, water quality changes in the subsurface actually occur, as well as the suitability of the recovered water for its intended purpose.

2.3 Aquifer hydraulics and the recovery of artificially recharged water

There are two physical characteristics which determine whether an aquifer is suitable for accepting artificially recharged water. They are the aquifer's permeability and storage capacity. A third important factor is the aquifer's hydraulic gradient, which relates mostly to the recovery of the recharged water. Key questions are:

- Is the aquifer sufficiently permeable to allow the recharge water to enter it?
- Does the aquifer have sufficient storage available to accept the water?
- Where should abstraction boreholes be located in order to maximise recovery efficiency - at the point of artificial recharge or down gradient from this area?

2.3.1 Aquifer hydraulics

Aquifers which are highly permeable and which have high storage capacities are more suitable for receiving additional recharge water than those which have low permeabilities and low storage capacities.

In fractured aquifers, the rate at which water will enter the aquifer will be determined by the permeability, extent and interconnectivity of the fracture system. In the case of basin infiltration, the success of recharge schemes is largely dependent upon the ability of the unsaturated zone to transmit water. In addition, the soil characteristics should be such that while the optimal infiltration rate is obtained, the contact time with the soil and aquifer material should also be sufficient for ensuring the desired quality improvements by sorption, degradation and other processes. Preventing full saturation below recharge basins will ensure maximum quality improvement.

In relation to aquifer storage, the main concern with Southern African aquifers, and in particular, secondary aquifers, is whether there is sufficient storage space to accept the artificially recharged water. On the one hand, aquifers are typically full during or shortly after the natural recharge periods, that is, when there is surplus surface water; and on the other hand, when there is little surplus water available for artificial recharge, during the dry months, they usually have available storage. Thus the suitability of the aquifer to receive additional water and the availability of water will often dictate whether a site is suitable for artificial recharge.

Where aquifers have been over-pumped (where abstraction is greater than natural recharge), space will be available for artificial recharge. Artificial recharge, however, should not only be considered as a means to replenishing over-abstracted aquifers, it should also be considered pro-actively as a means of maximising the use of available aquifer storage. In this respect, if a dependable source of suitable quality water is available for artificial recharge, then it would be worth considering "over-pumping" the aquifer, and on a regular basis, replenishing it with artificially recharged water. In this way, the use of available surface water, groundwater and sub-surface storage could be maximised.

The hydraulic factors that were considered at each study site are summarised below:

Windhoek

In order to establish the availability of space in the aquifer to accept large volumes of injected water, it was necessary to study historical water level, abstraction and rainfall data. This data showed that after years of high abstraction in the wellfields water levels dropped by tens of metres, and they took years to "recover". This showed that space was created in the wellfield areas, and that by injecting water into these areas, the aquifer could be rapidly replenished. In this way, the wellfield areas could be made to "recover" in a fraction of the time they would without artificial recharge.

On closer examination of water level data, it became apparent that even after years of resting the aquifer, the water levels never recovered fully to their pre-abstraction levels. Of particular concern was the steady decline in water levels in boreholes in the foothills of the mountains. This decline was seen to continue even when the aquifer was being rested, and it showed that the aquifer was being over-pumped, even though water levels appeared to "recover" in the wellfields. Thus over the past fifty years of pumping, a vast portion of the aquifer had been dewatered, and this created the opportunity to artificially recharge the main storage part of the aquifer, which are the quartzites that form the mountains.

Existing borehole records indicated whether the aquifer was sufficiently permeable to accept injected water at rates that would make the implementation of an artificial recharge scheme feasible. It was however necessary to establish the most permeable parts of the aquifer in the targeted injection areas. This was achieved by studying geological and fracture trace maps, and by developing a new lineament map that combined historical data with newly mapped fractures. The newly mapped fractures (mostly faults) were identified from a remote sensing study and field mapping.

Calvinia

The breccia pipe used in the artificial recharge study is hydrogeologically, fairly unique. It is about 100 m in diameter, and very poorly connected to the Karoo Sequence aquifer in which it is located. Water level recovery after abstraction is extremely slow - it took years to recover after the constant discharge test. For this reason, the groundwater compartment formed by the breccia pipe needs to be artificially recharged if it is to contribute on a sustainable basis to the town's water supplies.

Karkams

Like in Windhoek, abstraction from Karkams's granitic aquifer has caused water levels to drop considerably. But unlike Windhoek, due to this aquifer's low permeability, the water level decline is believed to be localised, and therefore artificial recharge close to or at the point of abstraction is necessary to rapidly replenish the aquifer. Prior to developing this artificial recharge scheme, it was necessary to study historical data in order to establish the effect that rainfall has on groundwater. The aquifer has a high, localised permeability along faults (~ 100 m^2/day), but it has a low storage capacity, and therefore natural recharge can replenish the limited storage space fairly rapidly. However, the rainfall is so low and variable, that in some years, the aquifer is not naturally recharged at all. This makes artificial recharge whenever surface runoff is available all the more important.

Polokwane (ex-Pietersburg)

The Polokwane infiltration system would not be effective if it were not for the expansive and deep (300 m wide and up to 25 m deep) alluvium that surrounds this non perennial river. The alluvium has the permeability and storage capacity to absorb water discharged into the river bed. It effectively acts as a sponge which releases water to the fractured gneissic aquifer below at the rate at which the fractures in the gneisses can accept the water. If this alluvium did not exist, the gneisses would only receive a small percentage of the municipal waste water that currently reaches the aquifer. The bulk of it would rapidly flow down the river and be lost for recharge at this point.

2.3.2 The recovery of artificially recharged water

The recovery of artificially recharged water from a fractured aquifer is mostly of economic concern as pretreatment costs prior to injection could be high. Where the characteristics and extent of the aquifer are known in sufficient detail, water levels can be managed to prevent losses of injected water. Theoretical recoveries can be determined by mathematical simulation but in practice water quality is mostly used for calculating the actual recovery. This can only be applied if one or more chemical parameters in the recharged water differ significantly from that in the aquifer.

Where the artificial recharge scheme affects only a part of the aquifer which is centrally located and hydraulically up-gradient of the production well field, losses of recharged water from the aquifer are, in general, regarded as being insignificant. This is the case in Windhoek. One injection area will be in the middle of the largest wellfield and therefore in the middle of the hydraulic depression; and the main artificial recharge area will be upgradient of the wellfields, in the foothills of the Auas Mountains.

Even though losses of injected water are likely to be minimal, it was still necessary to develop an understanding of the aquifer's discharge area. The Windhoek aquifer is bounded in the discharge area by relatively impermeable schists - which is the reason springs flowed in the schist terrain when groundwater levels were high. In order to determine the potential losses through the schists, the hydraulic parameters of schist boreholes were established by undertaking pumping tests. This included boreholes that penetrate both major and minor fault zones. From assessing the permeability of the aquifer's discharge area, it became evident that very little water can be lost from the system. Nevertheless, a mathematical simulation model is being developed for managing abstraction and recharge of the aquifer.

The Karkams site has similarities to the Windhoek site, in that it may be necessary to induce flow towards abstraction boreholes by pumping either the injection boreholes or boreholes down gradient of the injection site. In Karkams, it was possible, from geological mapping and pumping test analyses, to establish that the flow of injected water would be along a fault zone; and abstraction from the injection site would reverse the hydraulic gradient thereby allowing the injected water to flow back to the point of abstraction. Any losses from this capture zone would be intercepted by another production borehole 2 km down-gradient of the injection area.

The Karkams site differs from the Windhoek site in that the injected water is of significantly lower salinity compared to the groundwater. The more the natural groundwater could be displaced the better the quality of the abstracted water will be. The method used evaluate the effectiveness of fresh water injection into a saline aquifer falls into the concept of *recovery efficiency*, which is discussed below.

In Calvinia, the breccia pipe is hydraulically, virtually separate from the surrounding Karoo Sequence aquifer. This was evident after the constant discharge test, since water levels in boreholes immediately outside the breccia pipe barely responded to abstraction from the breccia pipe. All injected water is considered to be recoverable, since leakage from the breccia pipe, even when full, is negligible. What is of greater importance in terms of using recovered water, is the degree of blending between fresh, injected water and old, non-potable groundwater, as well as the water-rock interactions that takes place in the breccia pipe. These concerns commonly fall into the concept of recovery efficiency.

Recovery efficiency is of concern in borehole injection schemes where the quality of the recharge water and the native groundwater are of vastly different. In the case of aquifer storage and recovery systems (where the same boreholes are used for injection and recovery), recovery efficiencies are most often defined in terms of water quality, for example electrical conductivity. The limit for the abstracted water is generally set at a higher salinity than that of the injected water. This reflects the typical current practice of injecting fresh water into saline aquifers. What is being abstracted is therefore not only the injected water but a mixture of injected and native water. This is continued until the proportion of native saline water becomes unacceptable.

Typical recovery efficiencies in aquifer storage and recovery systems are found to be up to 70 percent. However, most schemes can be developed to 100 percent. Exceptions are very transmissive, highly saline aquifers which seldom reach recovery efficiencies in excess of 70-80 percent.

When considering an artificial recharge facility, the recovery of recharged water should be compared to the efficiency of surface storage facilities, such as dams, which lose vast volumes of water to evaporation and leakage.

2.4 Economics

Unused aquifer storage capacity can be developed at a significantly lower cost than surface storage facilities, and without the adverse environmental consequences frequently associated with surface storage. Often the overall costs of artificial recharge operations are less than half the capital cost of conventional water supply alternatives, especially those involving development of new reservoirs, treatment facilities or extensive pipelines (National Research Council, 1994).

It is important when undertaking a cost benefit analysis on various water supply options to ensure that all the costs are considered. These should not only include the costs associated with developing and operating the schemes, but also the savings from minimising water losses. Because evaporation losses are so high in arid and semi-arid areas and because seasonal rainfall results in large quantities of river water being lost to the sea, in many cases it will be cost effective to store water below the ground.

A detailed economic feasibility study is being undertaken in Windhoek in order to establish the scale of the Windhoek artificial recharge facility. Previous estimates have shown that the scheme will be highly cost effective, and about five times cheaper than taking water from the Okavango River (Van der Merwe, 2001; Murray, *et al*, 2000).

The value of having a dependable quantity of water is usually considered to be of prime importance. In the case of Windhoek, the security of having an additional 25 - 30 Mm^3 in the city's water bank within five years would be of great value. Likewise in Calvinia, having an additional three months supplies in their water bank is considered to be a vital supplement to the town's vulnerable surface water resources.

In the case of the other pilot artificial recharge schemes the economic benefits have not been assessed and this will only be done once several recharge cycles have been completed. In these cases, the availability of potable water far outweighs any economic considerations.

2.5 Management of the scheme

In general, artificial recharge schemes commonly involve surface or waste water capture, treatment, pumping, distribution and water quality monitoring. In order for these processes to be efficient, careful planning and management is needed. One of the key management functions is to minimise clogging. In the case of *surface infiltration systems*, a wetting and drying cycle with periodic cleaning of the bottom is used to reduce clogging by accumulated suspended material. It is also essential to maintain the quality of the recharge water.

In *subsurface infiltration systems*, like pits and trenches, it is essential that poor quality water be kept out of the infiltration galleries. The key management tasks include monitoring the quality of the recharge water, and maintaining good quality recharge water.

The potential for clogging is especially high in *borehole injection systems*. For this reason it is essential that the high quality of the recharge water be maintained. The key management issues are: maintaining the water treatment works which supplies recharge water; monitoring the recharge water quality; monitoring injection rates (injection efficiency); and restoring injection efficiency by backflushing, and other methods.

In order to maximise the efficiency of a large scale artificial recharge scheme, it is necessary to have the services of a full-time person with a good understanding of the aquifer. This is particularly important when clogging prevention and management is a key concern. All schemes will run into problems or lose their efficiency if the responsibility and tasks required to manage the schemes are not clearly defined.

In the case of small scale schemes pertaining to community water supplies, it may in certain instances, for example at Karkams, be possible to design the scheme for minimal water treatment and maintenance. Nevertheless, some monitoring and management on a regular basis remains essential for ensuring the success of the scheme.

CHAPTER 3:

Conclusions

Contents

3. CONCLUSIONS

The objectives of this assessment into artificial recharge to secondary aquifers, as set out in section 1.2 of this report, have been reached.

Objective 1

To test the artificial recharge concept in South African secondary aquifers.

Suitable secondary aquifers and test sites were identified and the artificial recharge concept tested in different situations. In particular, the large-scale Windhoek pilot study and the very small scale Karkams application confirmed the wide range of situations where the technique can succeed. This is of importance to large towns with extensive infrastructure and skilled staff, and to rural communities with minimal infrastructure and human resources.

The associated research product of one or more operational artificial recharge site could be delivered as the three sites Karkams, Windhoek and Calvinia are fully operational.

Objective 2

To demonstrate the potential for artificial recharge in South African secondary aquifers to hydrogeologists and water resource planners.

The technical success of the Windhoek artificial recharge pilot studies is confirmed on a management level by the fact that the Windhoek Municipality has decided to continue with the full-scale implementation of artificial recharge. This is to be implemented in the recommended phased approach set out in Section 1 of this report. The next phase of implementation is being funded by the Windhoek Municipality and is outside the scope of this project.

The results of the injection tests in Windhoek and Karkams were presented at seminars to the Groundwater Division of the Geological Society of South Africa, at a number of national and international conferences and at South African universities (Appendix 1). In addition to these presentations, the Karkams and Windhoek sites have been visited by a number of students, hydrogeologists and water resource managers. The case studies in sections 1 to 4 of this report cover the third research product which is to report on experience gained from the test sites.

Objective 3

To train local water resource managers in the operation and management of the pilot artificial recharge schemes.

During the pilot injection tests in Windhoek, Calvinia and Karkams, local water resource managers and operators were trained in the on-going operation and maintenance of the schemes. The second research product which entailed developing artificial recharge operation and management guidelines [developed from the pilot site(s)] was met in the case of Karkams, where a complete operating manual was compiled (Section 3, Appendix 3.4). This was the only site where three full injection runs were achieved and where it was possible to develop specific operational guidelines. Generic lessons learnt from all pilot study areas and general operating principles are described in Chapter 2 of this report.

This study was undertaken at an opportune time - the Windhoek municipality had recently completed a brief trial borehole injection test, and was ready to do more pilot injection tests, and the Calvinia municipality had been looking for options to increase the reliability of their water supplies. Both municipalities welcomed the opportunity to participate in this pilot artificial recharge study.

Karkams now has an effective artificial recharge scheme - even though it is a small-scale operation with a maximum injection rate of 1 L/s. This year (2001) saw the completion of its third controlled injection run since this project started. The Windhoek municipality is currently implementing a large-scale borehole injection scheme with a planned injection rate of 200 L/s. The Polokwane waste water infiltration scheme remains a successful system of waste water management and re-use. The only site where it is premature to conclude the success of the scheme is at Calvinia, where more abstraction-injection runs are required before the value of this scheme will be known. At the time of writing this report, the Calvinia municipality is busy with their second abstraction-injection run.

The concept of applying artificial recharge technologies in Southern African secondary aquifers has been demonstrated. A number of groundwater scientists and engineers have visited the sites, and fourteen papers based on this study have been presented at local and international conferences. What remains is for this technology to be implemented wherever the conditions are suitable in this water-strapped region of Africa.

Internationally, the concept of artificial recharge in aquifer management and in integrated water resource management is well established. Previously South Africa has only applied this technology to a primary, unconsolidated sands aquifer, namely the Atlantis aquifer. The pilot studies of this project have demonstrated that this technology can successfully be transferred to Southern Africa's predominantly fractured, hard-rock aquifers.

The success of the borehole injection tests in Windhoek and Karkams, the positive effect of infiltrating treated waste water in the Polokwane gneissic aquifer, and the detailed geochemical study that was required in Calvinia, have added another dimension to the fields of hydrogeology and integrated water resource management in Southern Africa.

CHAPTER 4:

Recommendations

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4. RECOMMENDATIONS

Specific recommendations pertaining to the particular pilot study are made at the end of each report (see Sections 1 - 4). Below are general recommendations regarding the development and promotion of artificial recharge in South Africa:

- The successful application of artificial recharge to both primary and secondary aquifers has been demonstrated and consideration should be given to using this technology in the following circumstances:
  - In areas where groundwater is being abstracted over and above natural recharge;
  - In areas with surplus seasonal surface water where groundwater abstraction is in balance with natural recharge, but where more water is required throughout the year;
  - In areas where groundwater is saline or non-potable;
  - In areas where a buffer zone may be required between good and poor quality groundwater (such as in the mining industry, where the migration of contaminated groundwater needs to be controlled);
  - In primary aquifers suitable for waste water discharge - like the Polokwane case. In these circumstances it is essential to treat the waste water to acceptable standards prior to artificial recharge.

- Strategic, large aquifers should be identified, such as the Table Mountain Group aquifers and the dolomitic aquifers, where conjunctive use between surface water and groundwater could be optimised. By utilising large aquifers in this manner, vast quantities of surface water that would either be lost by evaporation or flowing into the sea, could be stored and used on an annual basis, or reserved for emergency supplies. Detailed research would be required to assess the suitability of the target aquifers. For example, in the dolomites, water compatibility and sinkhole formation would need to be studied.

- This technology should be introduced in tertiary educational institutions. Water resource planners, water engineers and hydrogeologists should be taught the value of this technology and the key factors that influence its success.

- The value of artificial recharge should be conveyed to a wide audience who are currently involved in water resource planning. The audience should include planners involved in all areas of water supply and aquifer management, that is, municipal, rural, agricultural, industrial and mining. Artificial recharge needs to be viewed as an integral part of sustainable groundwater management and should be considered as a potential option in all groundwater development plans.

- In order to convey the artificial recharge technology, lectures throughout the country and demonstration site visits should be carried out. Supporting notes or booklets on theory, planning and case studies should form part of this technology transfer package.

- The Polokwane study needs follow-up research in order to quantify more accurately the effect of waste water infiltration into the gneissic hard-rock aquifer. Important data is currently unavailable. This data should be obtained, assessed and integrated with current knowledge prior to presenting this infiltration scheme as a model for waste water discharge - aquifer recharge schemes.

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APPENDIX 1:

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APPENDIX 2:

Review of Project by Dr P Dillon

PILOT ARTIFICIAL RECHARGE SCHEMES: TESTING SUSTAINABLE WATER RESOURCE DEVELOPMENT IN SECONDARY AQUIFERS

Project Review commissioned by Ricky Murray (CSIR, Stellenbosch)
undertaken by Peter Dillon (CSIRO, Adelaide)
21-23 November 2000

These are comments arising from a reading of the May 2000 Progress report to the steering committee of the Water Research Commission, and from presentations given by Ricky Murray and the following scientists on 21-22 Nov 2000, at CSIR in Stellenbosch, attended by Peter Dillon and (for the water quality components) by Gideon Tredoux.

These project presentations were given by Ricky Murray and for each of the project sites where artificial recharge of groundwater took place also by the following staff:

Karkams	-	Alan Hon
Calvinia	-	Lisa Cave
Pietersburg	-	Irene Saayman
Windhoek	-	Ricky Murray

Three sites involved investigations of artificial recharge by well injection in fractured rock and the remaining site, at Pietersburg, involved infiltration of reclaimed water discharged to a streambed.

Karkams

The investigations involving injection into a well in a fault zone from water harvested from a small creek in which a sand filtration bed was installed. Monitoring wells facilitated interpretation of the technical viability of the operation and of the fate of injected water in the aquifer.

This operation has been an unqualified success. There is no doubt that this has caused a substantial improvement in the quality of water abstracted from an adjacent town water supply well, has increased groundwater storage. There is adequate evidence to indicate that the rate of groundwater abstraction since 1996 could not be regarded as sustainable in the long term without artificial recharge.

The low cost of the treatment, absence of any indication of clogging, high injection rates with respect to abstraction rates, (contrary to that observed in sedimentary formations, and thought to be due to hydraulic connection with fractures normally above the piezometric surface), and excellent hydraulic connection with observation wells along the fault (from pressure and EC monitoring) suggest that this is a very suitable form of resource enhancement.

An issue yet to be addressed is the potential for increased concentrations in arsenic due to submergence and dissolution of oxidised sulphidic material (which could in some places contain arsenopyrites) along fractures which have become unsaturated due to drawdown of the groundwater storage. It is recommended that some monitoring of arsenic in recovered water quality, ambient groundwater and injectant be undertaken to explore this potential, at this site and at least once at each new site where artificial recharge is undertaken. This is unlikely to be a problem, but for public and scientific reassurace, some monitoring could lay this issue to rest, or suggest remedial strategies.

Due to rapid transport of injected water to the abstraction well, it is recommended that the current practice of disinfecting the water before use in water supply is continued. In the event of failure of the disinfection system, the risk of presence of pathogens in the water supply would be increased as there is very little residence time for microbial attenuation in the aquifer between recharge and recovery, which is in contrast with natural recharge to this confined groundwater system.

This suggests that for widespread applications it would be highly desirable to undertake some studies of the subsurface attenuation of pathogens of most concern, so that some experience is accumulated in evaluating the risks to water supplies. This could lead to the opportunity to avoid the need for a disinfection plant for some smaller water supply systems, say for isolated farms, where residence times between injection and recovery are demonstrated to be longer, or can be controlled, for example where there are two water supply wells, and the one nearest the recharge well is not used for a specified period following injection. The Karkams site is a difficult one to perform these microbial evaluations, and it may be desirable to establish microbial (bacterial, protozoa, viruses, helminths) attenuation experiments, say using diffusion cells, at more conveniently accessible sites considered to be representative of the aquifers in those areas.

It is highly desirable to continue the experimental injection and abstraction for at least another year to monitor the salinity increase during abstraction between the injection periods. As seen at a site in South Australia, the salinity increase was slower following each successive year of injection, and this may be used to infer the recovery of injected water and the possibility for scavenging wells further down-gradient along the fault.

A hydrologic evaluation of the surface water yield warrants consideration, possibly in a sequel project, to estimate the potential mean annual recharge, and the need for further sand filters and pipes to other injection wells, and to determine the possible effects of drought on system performance. This evaluation should consider the impacts of recharge diversions on the downstream watercourse and ecosystems. The ultimate scale of scheme and management options to improve EC in the supplied blend of water from abstraction wells, especially in drought conditions, warrant evaluation.

In my view an economic evaluation of the artificial recharge scheme should be performed, but not necessarily as part of the current project, to evaluate the options considered above. The costs of alternatives to the artificial recharge scheme, eg pipelines to other water sources or costs for reverse osmosis and brine disposal, could be documented, and compared with the costs of artificial recharge.

Following the extended monitoring period, it is recommended that the Karkams artificial recharge scheme be documented in a paper at the 4th International Symposium on Artificial Recharge, and be abstracted into publications which are widely distributed in the water supply industry of southern Africa.

Calvinia

This site is the most contentious and in some ways the most intriguing of those studied. The breccia tube was chosen as a storage site due to its excellent hydraulic confinement. Its drawback is that it contains water of high pH, fluoride and arsenic content, that exceed drinking water criteria. The notion is to pump out this water and replace it with better quality surface water and then use this as a drought and emergency supply.

Rightly, in my opinion, the work has focussed on the water quality issue, and attempts to evaluate via laboratory experiments and geochemical equilibrium modelling, the likely water quality of the recovered water. This work formed the basis of Lisa Cave's MSc thesis, and was rigorously performed.

Laboratory experimentation was performed under aerobic conditions with batches of various rock/water ratios, crushed and powdered rock, and mineralised (fracture surfaces) and matrix (away from fractures) material. Results indicated that all factors were relevant to the quality of equilibriating water. The geochemical modelling represented the effects of various proportions of injectant and ambient water on the resultant water quality for a given host rock taken to be representative of the breccia.

However there remains a gap between use of these results and an ability to predict the change in quality of water during storage of surface water in the breccia, including some imponderables that can only be solved by a field experiment. It is recommended that a field injection proceed at a time when it can proceed quickly with a significant volume of water (to avoid confounding of results by having water of a wide range of ages with respect to the total storage period). The field injection and storage experiment will require water quality monitoring that allows some interpretation of the processes and it is strongly recommended that organic carbon, dissolved inorganic carbon, nitrogen, phosphorus, sulfur-34, C-14, and C-13 be included in the monitoring program. Oxidation of injected organic carbon can increase the partial pressure of CO2, and hence changes acid-base and redox reactions within the aquifer, and consequently shifts the equilibrium products.

In preparation for the field injection, it is recommended that two laboratory column experiments be performed with subsamples of the same crushed (but not powdered) rock. (Mineralogy of the remaining subsample would be evaluated.) These column studies would be undertaken under anaerobic and intermittently aerobic conditions to represent conditions in the permanently and intermittently saturated zones respectively. It is suggested that continuously recirculating water be cycled through each column (with an anaerobic reservoir to allow water sampling and EC and DO probes) until water quality parameters become stable. Each column should be initiated by running breccia groundwater through until water quality is stable. One column should then be allowed to drain and be exposed to air (for a minimum of 3 weeks). Then surface water would be used to displace air and groundwater and recirculated through both columns until equilibrium, and water quality parameters measured, including isotopes. The aerobic column would then be drained and left exposed to air, while the other column would be purged with native breccia groundwater, until the experiment was repeated by recommencing the circulation of surface water through each column. The anaerobic column would need to be maintained under anaerobic conditions at all times (representing conditions in the aquifer beyond the immediate vicinity of the injection well). It is expected that each of the column phases should be at least 3 weeks to allow for microbial populations to grow under each new set of environmental conditions, as micro-organisms may mediate reactions that influence water quality, eg SO4 reduction. The experiment can be extended if successive injection and storage cycles give results that indicate trends. At the end of the experiment the mineralogy of column materials should be reanalysed, and a mass balance performed for at least As and F.

Such experiments do not answer the question on the degree of exposure of minerals participating in reactions and the relationship between this in the columns and that in the field. However this type of experiment is a step closer to providing forecasts for the field experiment, and may assist with interpretation of the field experiment, whether the results are similar or contrasting. They may also suggest solutions such as, use of packers to recover water from within a depth range that excludes recovery of deep predominantly native breccia groundwater and shallow water that may have been subject to oxidation and is possibly enriched in As and F.

A mass balance for As and F could possibly be estimated now and based on breccia mineralogy. This, together with the column experiments could assist in estimating the flushing volumes that will be required apriori. Even if these look large, it is suggested that the field experiment should proceed, as there are many uncertainties associated with the accessibility of the As and F for leaching.

It is recommended that consideration be given to water treatment alternatives for F and As removal, and if this requires experimentation, to establish a pilot plant if possible prior to recovery of the water to be stored in the breccia. This would enable this to have a beneficial use, in the event that it exceeded guideline values, and would facilitate further water being made available to the trial in the event that the first water recovered is not of suitable quality.

Some hydrologic evaluation is warranted on the additional water that this scheme would make available assuming that recovered water is fit for supply. This would be based on the estimated reduction in evaporation for water which otherwise would be stored in the dam or lost as spill. Some estimate of the economics of storage in the breccia tube is warranted under scenarios ranging from all water stored being available for recovery, to only a proportion of water being recoverable.

Pietersburg **(Now named Polokwane)**

The two questions posed had been answered from

(1) *does recharge occur from alluvium to underlying gneiss?*
(2) *what impact is there on groundwater quality?*

Na, Cl, TDS, EC and alkalinity all doubled downstream of the wastewater discharge, however all appeared to be in accordance with drinking water criteria, for the data shown for two wells.

Given the settlements adjacent the Sand River in the catchment upstream of Pietersburg, it is likely that bacteriological quality of runoff is similar to that of the wastewater.

Some EC data from profiles run in several wells showed that EC variations in alluvium and gneiss were significant, and this may also indicate that variations in other water quality parameters may be greater than for the bores shown, and that mixing ratios may be more variable. There were hints of sulphate reduction upstream but not downstream.

Water level contours were based on wells in or close to alluvium (and undifferentiated between alluvium and gneiss) and inferred directions of flow are only approximate. As expected these suggest a cone of depression beneath the alluvium, indicating bank filtration as opposed to a recharge mound. Emergency supply wells are in alluvium and gneiss, so attempts to estimate recharge to alluvium and gneiss independently are likely to be confounded, in spite of having estimates for permeability of the zone between alluvium and the first water cuts in the gneiss. .

There were no levels and water sample analyses from wells in the gneiss away from the river downstream of the wastewater discharge, nor from wells in alluvium upstream of the wastewater discharge. This makes rigorous interpretation for quantification of recharge difficult, and possibly confounds the use of Deuterium and O-18 measurements for estimating concentration of solutes by evapotranspitration, before dilution (?) with ambient groundwater in the gneiss. The similar chloride concentrations in wastewater and gneiss wells downstream of the discharge possibly reflect that dilution by ambient groundwater compensates for any increase due to enrichment by evapotranspiration.

It is proposed that recharge be estimated differently depending on whether the stream is flowing or not.

(a) If there is no flow, estimating the wetted area over which evapotranspiration takes place, and take a factor of say 0.5 to 0.7 of pan evaporation, to give the rate of evapotranspiration to deduct from the wastewater discharge rate to give a net recharge.

(b) If there is flow, determine the increase in stream head and wetted area in the study zone and use this with an analytical model for recharge depending on whether the alluvial aquifer is hydraulically connected or disconnected from the stream, to calculate the incremental recharge attributable to wastewater discharge. Can use the permeability or leakage rate derived from (a) to parameterise the model in (b).

Look at the historical flow record to accumulate these over the years when you wish to make these estimates.

Chloride values of reclaimed water (allowing for E/T enrichment), ambient groundwater, and possibly (if known) typical river water at times when it contributes recharge, could also be used to infer proportion of total recharge to alluvium from reclaimed water. This will require knowledge of typical river water chloride concentrations at times of high flow. Perhaps if there were bores in alluvium upstream of the reclaimed water affected area this could be determined in a more time-integrated manner. What is unknown is the effect of pumping from both alluvium and gneiss for water supplies, and the possible inducement of flow from gneiss into alluvium as a result. Water quality data could be used together with estimated recharge rates to infer mixing ratios at observation wells in gneiss. In the absence of water samples from gneiss downstream of reclaimed water affected zone and further away from the alluvium, it is assumed that these may be represented by samples from wells upstream of the discharge point.

Clearly this Pietersburg example of artificial recharge is a case of water reuse where riverbank filtration, natural and induced is used to reclaim water for emergency town water supplies.

A limitation of the study is the inability to obtain water quality samples from individual wells. This could be overcome by the municipal water authority installing taps at each well head.

The wastewater treatment plant appears to have been well-operated, as nitrate concentrations are very low. It would be reassuring to obtain phosphorus, total nitrogen and ammoniacal nitrogen in addition to oxidised nitrogen to complete the nutrient story for this site. The effects of effluent discharge on stream water quality and biota are outside the scope of this study, but some reference could be made to observed effects.

If water quality continues to be well-controlled by the water authority, and recharge is found to be a small component of discharge, this site could be promoted as an excellent example of water conservation and reuse of water (bearing in mind earlier comments on the increase in salinity of the alluvium water). If recharge efficiency is low, and there is adequate storage in the aquifer, subsurface slotted drains could be used where alluvium is permeable and well connected to underlying gneiss to improve recharge efficiency.

Windhoek

This study has been well executed and has generated data on flow and solute transport in faulted fractured rock aquifers that warrant further evaluation. Results have been extremely successful and will stand out as an internationally acclaimed demonstration of artificial recharge. When fully developed an artificial recharge project here is likely to be capable of providing up to 20% of Windhoeks water supply and substantially increasing its emergency reserves.

The study has been conducted rigorously within the constraints of a full-scale field operation, and has produced excellent data on pressure variations in the aquifer as a result of injection and abstraction, to enable the connectivity of bores via fracture systems to be evaluated. It has also yielded excellent data on electrical conductivity changes at observation wells in response to injection of the fresher surface water, and some startling results from a fluorescein slug tracer test. This information requires assimilation and development into a conceptual model of the behaviour of the aquifer system, which would significantly extend the scope of the current project. This would provide a basis for a groundwater management model of the aquifer system taking account of injection and abstraction regimes to enable restoration and enhancement of the currently depleted groundwater system. It would enable development of scenarios for developing the groundwater resource to most efficiently improve the quantity and quality of recovered water, and to understand the constraints on system development and the trade-offs between expansion of annual abstraction and expansion of groundwater reserves.

Water quality issues yet to be evaluated are: As, Radon and possibly DBP's (notably THM's). It is recommended that some preliminary monitoring of these parameters be included in project development to rule these out or to determine whether remedial actions, such as further treatment of injectant or recovered water are required. These could be related to an international project currently underway entitled 'Water Quality Improvements During Aquifer Storage and Recovery' on a data and information exchange basis.

We discussed the need for GAC treatment and rechlorination. It is recommended that DOC (and TOC) be kept to as low a level as possible in injectant, in order to minimise microbial growth in the aquifer and thereby reduce production of biomass and extra-cellular polysaccharides (slimes) that could ultimately clog the flow pathways especially where these have small apertures. It also reduces the substrate available to support microbially mediated reactions, including those redox-related reactions which could influence water quality, such as sulphate reduction. Given the cost of GAC treatment and rechlorination, a long-term trial undertaken with and without this treatment train may be warranted. Such a trial would demand the presence of at least one observation well in close proximity to the injection well, in order to assess any increase in well losses which may be attributed to microbial action.

Pathogen concentrations in the recovered water are unlikely to be a greater issue than for the existing groundwater abstraction bores as the water injected water already meets the required microbial quality requirements for drinking water.

Summary

Of the four sites studies, two are seen as major successes; Karkams and Windhoek. These projects, demonstrate at small and large scales respectively, a significant improvement in both the quantity and quality (reduced salinity) of water supplies. Potential for reversing declines in groundwater storages, expansion in water supplies, and in increasing security of supply are clearly evident at both sites. The technology has been decisively proven effective at these sites. Further monitoring at each site would be an excellent investment to comprehensively demonstrate that these injection systems are sustainable. Some additional water quality evaluations, notably of, arsenic, fluoride, radon, THM's (Windhoek only) and organic carbon, are necessary to provide reassurance that these potential water quality issues are addressed. Bacterial quality assessment of recovered water is not required where the recovered water is disinfected. However if in future it is intended to apply Karkams-like schemes for small village water supplies, it is recommended that research on pathogen survival and transport be undertaken to understand the enhanced risks attributed to aquifer injection in the event of failure of the water supply disinfection system.

The Calvinia site is problematic, and while the hydraulic containment of the breccia tube is excellent, the geochemical considerations with respect to arsenic, fluoride and pH warrant further attention. It is suggested that recirculating column experiments under anaerobic conditions be undertaken to complement the existing aerobic batch studies, to assist in predicting the quality of water during storage and recovery. This has some scientific as well as practical merit. A full scale injection should be attempted when water can be injected continuously at high rate into the storage, to minimise the time of filling the evacuated storage.

It is suggested that an associated experiment on treatment methods to reduce As, F and pH to drinking water guidelines be attempted in parallel, so that in the event of recovered water not meeting the guidelines it need not be discharged to waste, and cycles of filling and emptying the storage can be achieved if results indicate that further flushing will result in recovery of suitable quality water.

The Pietersburg site represents a wastewater reuse application, involving bank filtration, which appears to be sustainable. The project lacks some key data required for estimating recharge rates but it appears at this stage this is a significant proportion of the total discharge and may locally enhance this by up to 40 or 50%. Natural tracer chemistry, notably use of chloride is more likely to give useful quantitative estimates or recharge than hydraulic modelling, but some basic water balance calculations may assist. Spatial variability of data tends to indicate that the system is more complex than that described by comparing water quality results from only a small subset of bores. Additional wells set back from the Sand River downstream of the effluent discharge point and in the alluvium upstream of the zone that is influenced by reclaimed water discharge would assist in giving more quantitative conclusions.

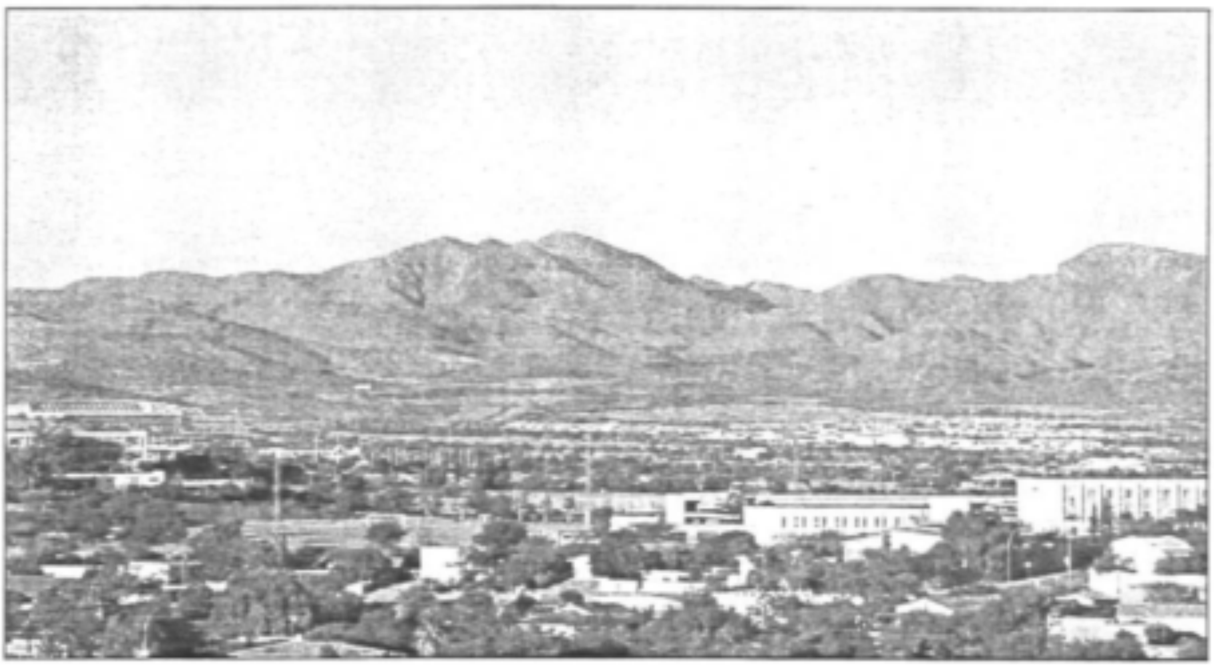
Overall, the project has addressed the major hydraulic and geochemical issues appropriately. Resources committed to each activity have been in balance with the expected benefits, and have been refined appropriately as the project has progressed. Project management has been excellent. Pietersburg has presented problems but these are being overcome. The success of two of the sites, Windhoek and Karkams can not be overstated. The benefits (water quantity, quality and security) for these communities alone will no doubt exceed the cost of the project many times over, and the value of these as demonstration sites, which should be widely promoted, would significantly enhance uptake of artificial recharge at all scales in fractured rock aquifers in southern Africa, and possibly other arid areas with similar geology. This is a low cost technology and will be of great value in achieving South Africa's plan for enhancing water supplies to rural and remote communities. The projects demonstrate the need to tailor the technology to the specific local conditions. No doubt this leaves room for many more projects under circumstances which offer different challenges.

SECTION 1:

Windhoek Pilot Artificial Recharge Study

by

E C Murray & G Tredoux



The City of Windhoek looking over the Group 9 and 12 wellfields towards the Auas Mountains

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1. INTRODUCTION

Since 1997 the Windhoek Municipality has been testing the concept of artificially recharging the Windhoek quartzite aquifer by injecting surface water into boreholes. This fractured aquifer has been used as a water source by the municipality since the early 1920's. The aquifer is however vulnerable to over-pumping. Since large scale abstraction began in the 1950's, water levels in the aquifer at large, have declined by tens of metres.

During years of high groundwater abstraction, water levels in the wellfields drop by 40 m or more. Following this, after years of low abstraction, the water levels recover, although never to their original, pre-abstraction levels. This "recovery" takes place irrespective of whether recharge has occurred during the recovery period or not. Even in very poor rainfall years, the water levels in the wellfield areas continue to rise. This water level recovery is therefore believed to reflect the rate at which water from other parts of the aquifer can drain towards the hydraulic depressions in the wellfields.

The aim of artificially recharging the aquifer is both to rapidly replenish the wellfield areas which have been over-pumped, and to steadily replenish the main storage parts of the aquifer which supply water to the wellfields. The main source of water for the wellfield areas, and in particular the Group 9 wellfield area where three borehole injection tests have been done, is believed to be the quartzites below the Auas Mountains (Figure 1).

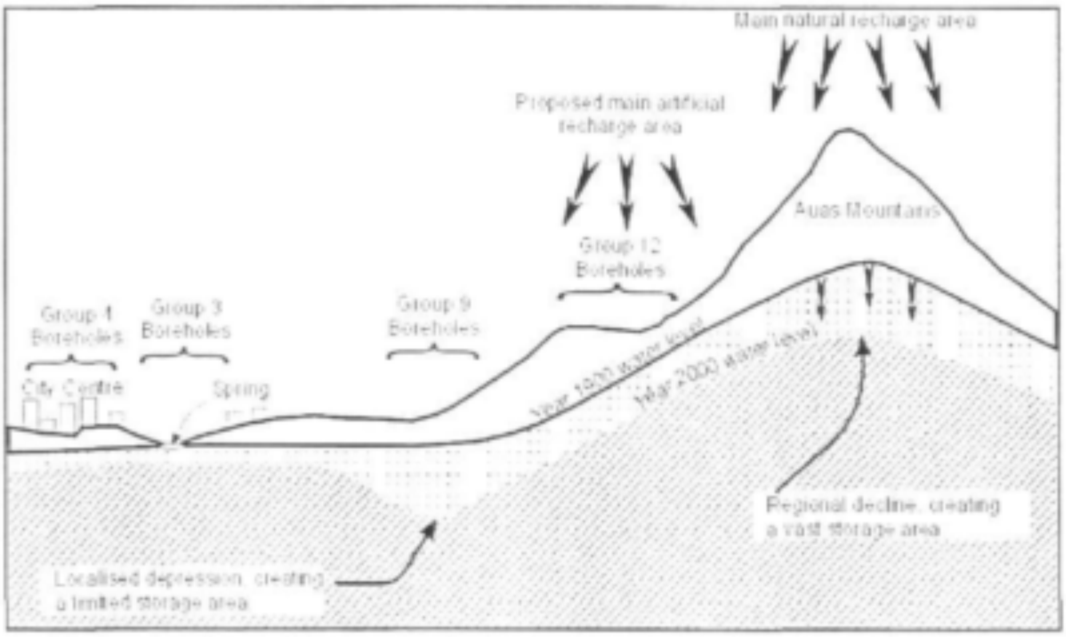


Figure 1 Hypothetical diagram showing postulated groundwater level changes since the commencement of abstraction, and the vast storage area under the Auas Mountains which has been reduced as a result of pumping

Windhoek's production boreholes are all located within a 10 km radius of the centre of the city (Figure 2). Between 1950 (when detailed records were kept) and the year 2000, the total volume of water pumped from the Windhoek aquifer was 103.4 Mm^3. This gives an average of 2.1 Mm^3/a. There have been four main periods of abstraction, and during these periods, which total 31 years, 86.3 Mm^3 or 83% of the total abstracted volume was pumped. The most intense abstraction period was between 1965 and 1970, where an average of 4.0 Mm^3/a was pumped (Table 1).

Table 1 Periods of high groundwater abstraction

Period	*Years*	*Total (Mm^3)*	*Average (Mm^3/a)*
1952 - 1962	11	27.5	2.5
1965 - 1970	6	23.7	4.0
1975 - 1983	9	20.0	2.2
1992 - 1996	5	15.1	3.0
		86.3	2.9

The annual water requirements for Windhoek over the past ten years ranged between 16 - 20 Mm^3. The periods of lowest usage, during the dry mid-1990's, correspond to the implementation of highly effective water demand management practices.

The drop in groundwater usage can be attributed to the completion of the city's supply dams: in 1963, the Goreangab Dam; in 1971, the Von Bach Dam (completed in 1970); and in 1984, the Swakopport Dam (completed in 1982). The high abstraction rates during the mid-1990's can be attributed to low rainfall and low water levels in Windhoek's three supply dams (the Von Bach, the Omatako and the Swakopport). Until the completion of the Von Bach Dam groundwater was the main source of water for the city (Figure 3).

The Group 9 wellfield, which forms part of the pilot artificial recharge study, accounts for 21.3 Mm^3 or 23.6 % of the total groundwater abstracted since 1950; and the more recently developed Group 12 wellfield, which was also targeted for pilot injection tests, accounts for only 2.4 Mm^3 of the total groundwater abstracted (Figure 4).

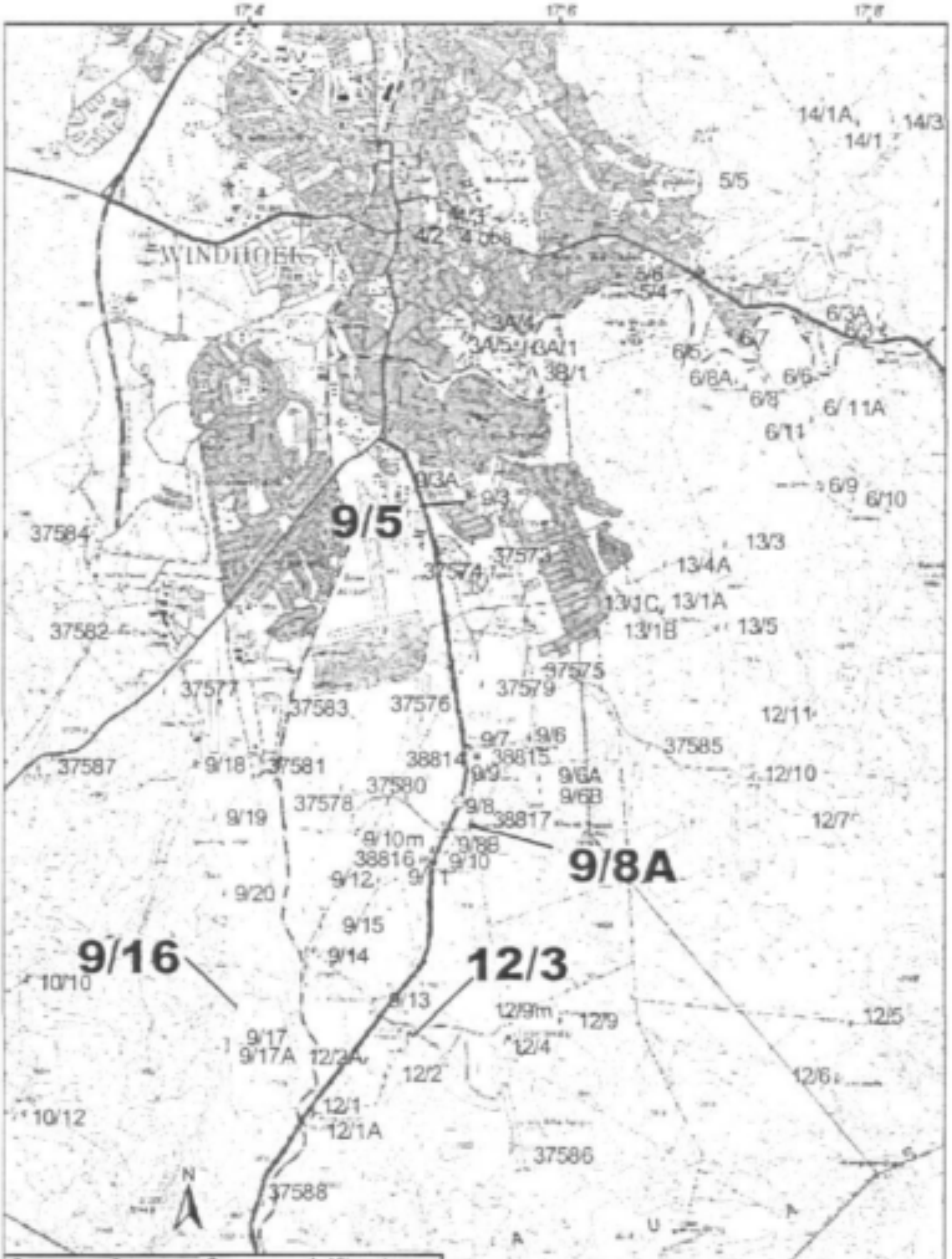


Figure 2 Borehole locations: Injection boreholes in bold (Source map published by the Surveyor-General, Windhoek, 1983)

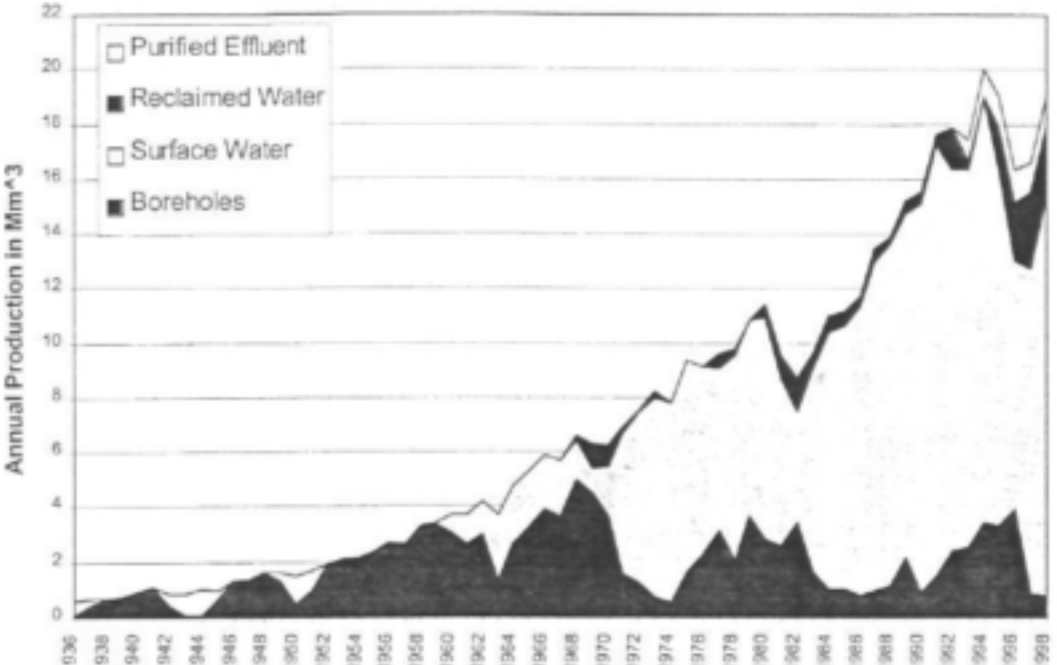


Figure 3 Windhoek's water sources, showing the heavy dependance on groundwater until the early 1970's when the Von Bach Dam came into production (Source: Van der Merwe, 1999)

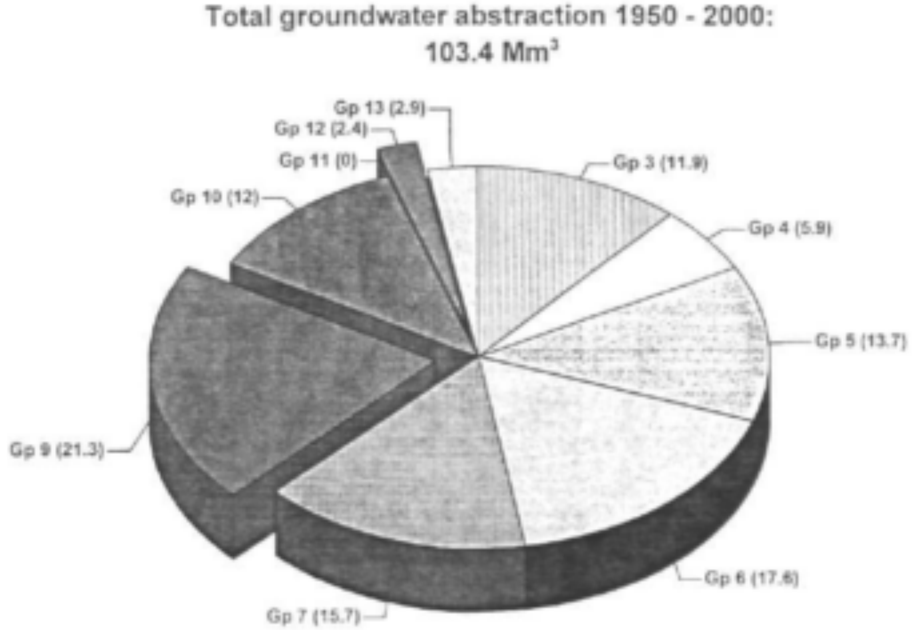


Figure 4 Groundwater abstraction per wellfield from 1950 to 2000. In brackets is abstraction in Mm^3

In all areas except the Group 3 and 4 wellfields, water levels have dropped since large-scale abstraction began. Figure 5 shows groundwater level fluctuations with time; and Figure 6 combines rainfall and total abstraction from all boreholes. The boreholes selected in Figure 5 represent the water level fluctuations in and around the Group 9 and 12 wellfield areas. Borehole 10/10 is a high lying borehole west of the Group 9 and 12 boreholes. It represents the water level trend away from the large scale abstraction of the Group 9 and 12 wellfields.

Excluding the Group 3 and 4 boreholes and the few boreholes that have perched water tables, the water levels do not return to their early, pre-abstraction levels after periods of rest. The general decline in water levels is attributed to abstraction being in excess of natural recharge. On average, the water level decline is 0.8 m/a. This is in response to an average abstraction of 2.1 Mm^3/a since 1950.

The fact that water levels have generally dropped since large scale abstraction began, provides the most compelling argument for artificially recharging the Windhoek aquifer. This is supported by the observation that in high-lying areas where abstraction is low, water levels are still seen to be dropping.

The boreholes that do not follow this trend (like the Group 3 and 4 boreholes, and boreholes 9/3 and 9/5) are located within the schists in the central part of the Windhoek graben. In these areas, the formation's permeability is low and flow towards any hydraulic depression is minimal. Even though deep, localised depressions form around pumping boreholes located on faults within the schists, the formation at large cannot yield the water to cause significant drainage from the area. This is in contrast to the more quartzose areas, where the higher formation permeability allows for general drainage from the area and therefore a more widespread hydraulic depression.

The Windhoek aquifer's sustainable yield has been estimated to be 1.93 Mm^3/a (CBA, 1999). Considering that an average yield of 2.1 Mm^3/a has resulted in a steady decline in groundwater levels in most areas, and in particular, the quartzitic high lying areas, it seems that 1.93 Mm^3/a may be far in excess of the aquifer's sustainable yield. Although no detailed natural recharge studies have been undertaken, the figure of 2 - 3 % of MAP has been suggested. Colvin (1994) suggested a figure of 2 % and Van Tonder (pers.comm.) obtained a figure of 2.7 % using various techniques. If a conservative figure of 2 % is used for the outcropping quartzites and micaceous quartzites (157 km^2), then the natural recharge estimate is 1.1 Mm^3/a. Assuming that this figure is a reasonable estimate, it would imply that the aquifer has been over-pumped during the past fifty years at a rate twice that of natural recharge. Irrespective of whether this recharge figure is correct or not, what is certain is that if the aquifer is used at the current average abstraction rate, it will continue to be steadily mined.

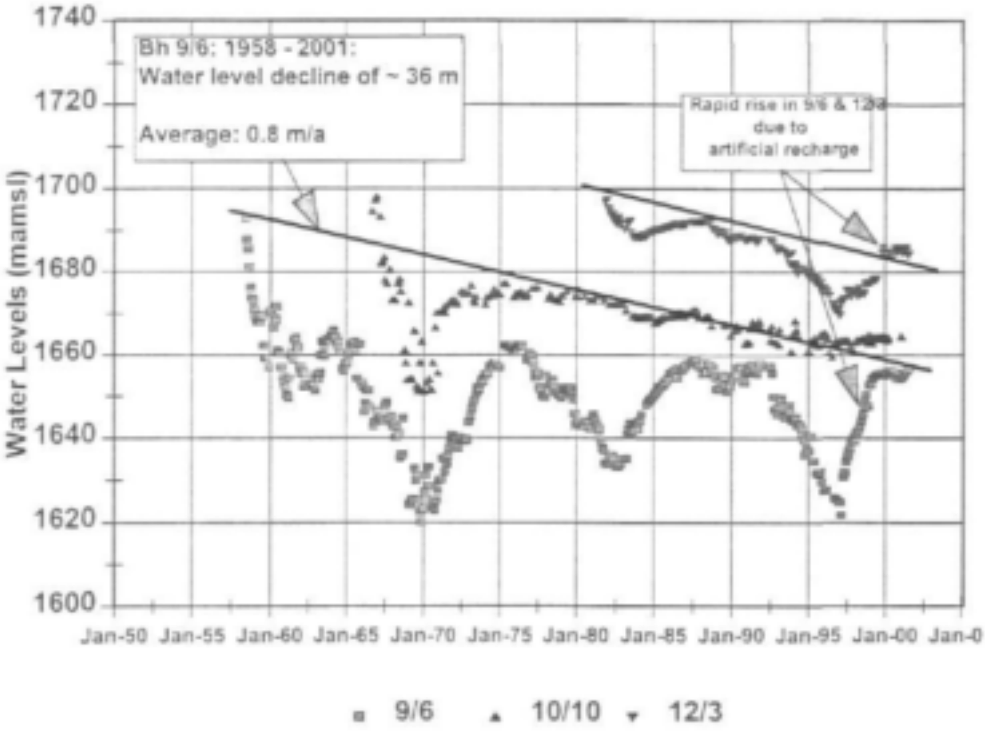


Figure 5 Boreholes 9/6, 10/10 and 12/3 historical water levels

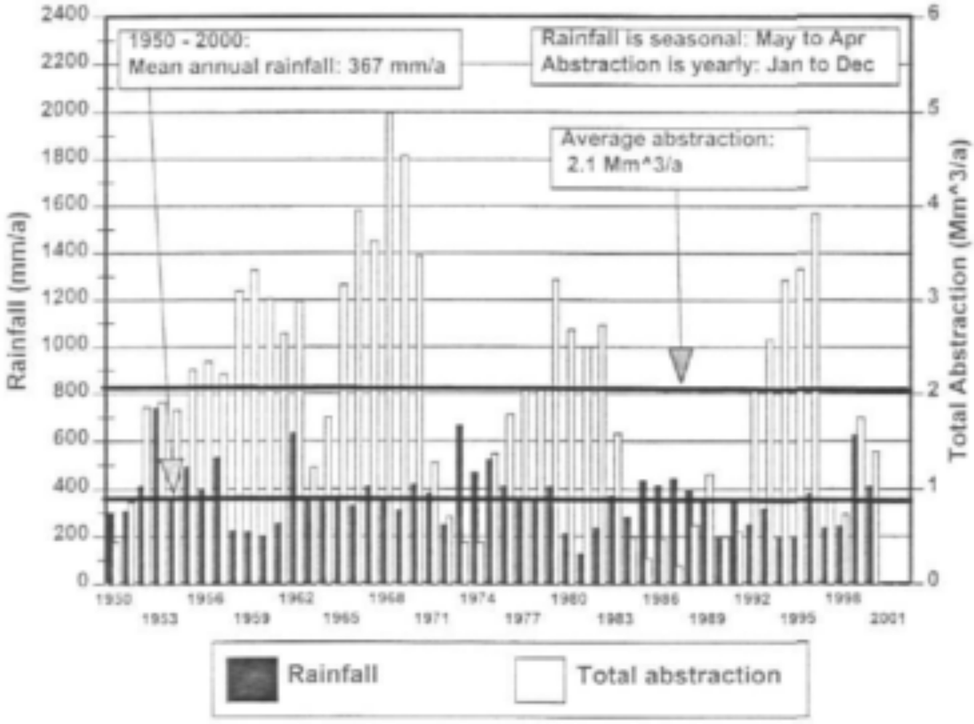


Figure 6 Rainfall and total abstraction from the Windhoek aquifer

Exactly what volume of water can be stored in the Windhoek aquifer is currently unknown, however, it appears to be in excess of 100 Mm^3. The 103.4 Mm^3 of groundwater pumped over the last 50 years is "old" groundwater, with ^{14}C-determined ages ranging between 3 100 to 24 400 years before present. It is therefore assumed that since abstraction on a large scale began, most of the groundwater has come from storage. This assumption is backed up by the 16 month pumping test done on borehole 9/14. During this test, which started in January 1973, and in which 368 000 m^3 was pumped, no significant changes were found in the ^{14}C content (Kirchner, 1976).

Existing information points to three pertinent observations:

i) The groundwater has, in all probability, been mined over the past few decades;
ii) In the aquifer south of Windhoek, a high proportion of storage occurs in the quartzites of the Auas Mountains;
iii) Abstraction from the Group 9 boreholes induces groundwater from the Auas Mountains towards the boreholes, thereby draining the water that is held in storage below the Auas Mountains.

Should these observations be correct, then the main area of artificial recharge should be as close to the Auas Mountains as possible (Figure 1). By recharging this area, the depleted storage under the mountain will gradually be replenished, and this artificially created hydraulic barrier will slow down the natural groundwater flow towards the town by "damming" the stored water below the mountain.

The overall objective of the artificial recharge project is to ensure a minimum level of supply from the Windhoek aquifer during periods of drought. This can only be achieved by continually artificially recharging the aquifer in the areas that have been worst affected by abstraction (the southern-most parts of the aquifer), and intensifying this after periods of high abstraction.

Four borehole injection tests have been completed to date. In 1997 an injection test was done on borehole 9/5 east of the Eros Airport (facilitated by Carr Barbour & Associates); and between July 1998 and September 1999 three borehole injection tests were done on the Group 9 and 12 boreholes south of the Eros Airport (facilitated by the CSIR). An overview if the four tests is given in Table 2.

Table 2 Pilot borehole injection tests

Injection Borehole	*Dates*	*Period (Days)*	*Injec-tion Rate (m^3/hr)*	*Total Volume Injected (m^3)*	*Treatment in Addition to the Standard Treatment*
9/5	28/1/97 - 18/2/97	21	10	5000	Chlorinated
9/8A	12/8/98 - 23/2/99	195	61	289000	Carbon filtration and chlorination
9/16	12/4/99 - 27/5/99	45	22	24000	Standard treatment only
12/3	10/8/99 - 14/9/99	35	118	99000	Standard treatment only

This report describes in detail the three borehole injection tests which were facilitated by the CSIR, and it briefly refers to the injection test on borehole 9/5 which was facilitated by Carr Barbour & Associates. The report also makes recommendations regarding the implementation of a full-scale artificial recharge scheme.

2. GEOHYDROLOGY OF THE WINDHOEK AQUIFER

2.1 Geological setting

The geology of the study area is extremely complex as a result of the geological processes associated with intracontinental rifting and continental convergence (Martin and Porada, 1977 and Hartnady, 1978). The Windhoek aquifer lies in the southern marginal zone that separates the Kalahari Craton in the south from the Congo Craton in the north. Formations within the study have been overturned in the process of orogenesis, and they have been subjected to a number of episodes of faulting including thrusting and rifting. The metasediments that form the Windhoek aquifer have their origin in the sedimentation that began about 1000 Ma ago with infilling of fault-bounded troughs in the rifted continental crust (Tankard, *et al*, 1982).

The southernmost part of the study area is dominated by gneisses and amphibolite schists of the Rietfontein Basement Complex which formed over 1000 Ma ago (Geological Survey, Windhoek, 1988). Thrust against these basement rocks, and overlying them are the metasediments of the Southern Marginal Zone of the Damara Sequence. They consist mainly of quartzites and schists and form the mountains in the south, east and west of the study area.

The Windhoek wellfield consists of interbedded, quartzites and schists of about 500 Ma which lie north of the Auas Mountains Range. These east-west striking metasediments dip 15° - 30° to the north and are overlain by sandy colluvium in the central, northern part of the study area.

2.2 Hydrogeological units

The hydrogeology of the area is dominated by the quartzite and schist horizons, and the faults and fractures that are prevalent throughout the Windhoek aquifer. The quartzites, being brittle, are highly fractured as a result of folding and faulting and have developed secondary porosity and permeability. The schists on the other hand are ductile and do not have well developed secondary permeability (Murray, 2001 and Carr Barbour and Associates, 1999). Both the schists and the quartzites are thought to have no primary porosity (Colvin, 1994).

The quartzites vary from being relatively pure in the south (the Auas Formation) to micaceous in the north (the Kleine Kuppe Member of the Kuiseb Formation). As the mica content, which is predominently the mineral sericite, increases in the quartzites, so the degree and intensity of fracturing decreases. Sericite is commonly referred to as a "sub-mica", since it is a mica-like substance that has characteristics the lie between the micas and clay minerals. Both have layer-lattice atomic structures. Sericite responds in a plastic or ductile manner to stress and for this reason, the openness of the faults and fractures in the more sericitic quartzites are less than in the more pure quartzites.

The schists are micaceous (predominantly sericite and biotite) schists, and due to their mineralogy and parallel arrangement of their minerals, are not conducive to groundwater flow. The same applies to the thin layers of amphibolite, with their orientated fabric and common association with thrusts. Like the schists, the amphibolites can be considered virtually impermeable except in areas cut by sizeable fractures.

The geology of the Windhoek aquifer can be divided, for hydrogeological purposes, into three main units of decreasing permeability: quartzites, micaceous quartzites and schists (Figure 7). A fourth unit, namely the gneissic material south of the Auas mountains, is not considered to have any effect on groundwater flow within the Windhoek basin. This unit is thought to be less permeable than the schists (Hälbich, pers. comm.), and is thus likely to act as a barrier to northerly flowing groundwater south of the Auas mountains.

Figure 7 Geohydrological units and prominent faults around the Windhoek graben (Murray, 2001a)

2.3 The Windhoek Graben Structure

The borehole injection tests described in this report were carried out on boreholes located at the edges of the Windhoek graben. It is not possible to accurately locate the faults that bound the Windhoek graben. On the western side of the graben, the roughly N-S line that marks the transition from the valley floor to the western hills indicates that the fault must lie in the vicinity of this line. Taking erosion into account, it seems likely that the Western Boundary Fault, lies slightly to the east of this topographical line. The prominent Venusberg Fault appears to form the Western Boundary Fault (Figure 7).

On the eastern side of the graben there is no clear N-S topographical line, and thus an estimate of the location Eastern Boundary Fault needs to be indirectly derived. From remote sensing evidence (satellite and air photo imagery) and surface geophysics in the borehole 9/8A area, it appears that an extension of the Bergtop Fault may form the Eastern Boundary Fault (Figure 7).

The bounding faults are also not necessarily single faults, but may well consist of complex fault zones of numerous sub-parallel faults that join and split from one another.

2.4 Secondary permeability

Owing to brecciation and fracturing along fault zones, the permeability of the faults is clearly far greater than that of the regionally jointed rocks. On surface, most N-S and NW-SE joints are open and can be expected to remain so to some extent to a few hundred metres depth (Halbich, 2000). The less prominent, widely spaced, E-W trending vertical strike joints are thought to have little regional hydraulic significance (Halbich, 2000). Based on field observations, Halbich (2000) estimated that for the pure quartzites and thick-bedded sericitic quartzites, the permeability of the breccia cum fracture zones could be 10 to 100 times greater than that of the regionally jointed rock, and that these near vertical conduits make up about 10% of the total rock volume.

Other sources of secondary permeability, besides faults and vertical joints, are joints and slip planes which developed in the quartzites during the period of low-angle thrusting. The hydraulic characteristics of these old structures are unknown, however, thrusting generally follows the least resistive formations like the pelitic rocks that through metamorphasim formed the schists. If this was the case in the Windhoek region, the permeability of the associated joints would be low. Associated with the thrust faults are thin layers of amphibolite. These poorly conductive layers and the quarts veins which tend to fill the thrusts would also impede groundwater movement.

Another source of secondary permeability within the quartzites may be bedding planes and sub-horizontal fractures. Like the thrust associated joints, no information exists on the permeability of these structures. Field observations of bedding planes and jointing parallel to bedding planes suggests that these features are not likely to be of great hydraulic significance.

The last source of secondary permeability within the quartzites is considered to have resulted from the breakdown, due to weathering, of the disseminated iron sulphides (Carr Barbour & Assoc, 1999). Drilling records reveal that such oxidation occurs to depths exceeding 100 m in areas where the water levels are lowest. This weathering may or may not have altered the permeability and porosity of the quartzites. Certainly in terms of permeability, this factor is not likely to be significant in comparison to the fault induced permeability.

In the less competent, mainly schistose rocks the permeability, even along most faults, is very low (Murray, 2001b, Halbich, 2000 and Carr Barbour & Assoc, 1999). The exception to this in the schistose rocks, are hydrothermally altered fault zones with considerable displacement, such as the Aretaregas Fault, the Pahl Fault and possibly the two boundary faults of the graben (Murray, 2001b and Halbich, 2000). Although the permeability of these faults may be low, these faults definitely breach the thick mica schists that form the southern part of the Windhoek aquifer.

The brittle, near-vertical faults can also be grouped into three main types of increasing permeability: faults with little or no associated hydrothermal alteration, faults with associated hydrothermal alteration and faults with post-hydrothermal alteration and brecciation. Although the latter fault type is generally considered to be the most permeable, infilling with siliceous material could plug these faults and leave them relatively impermeable. The low-angle thrust faults are believed to be impermeable except in areas where they are cut by near-vertical brittle faults.

2.5 Groundwater flow direction

The dominant groundwater flow direction in the Windhoek Basin is from the Auas Mountains in the south to the city in the north, and then towards the Aretaregas River in the north-west. This flow is dominated by the numerous faults and fracture zones that cut through the area. Figure 8 shows the current-day groundwater level contours (Murray and Tredoux, 2001). The widespread hydraulic depression that stretches from the Group 9 area in the south to the Group 6 area in the north-east is a result of large scale abstraction over the years. The hydraulic depression is clearly shown in the current-day, north-south cross section through the Group 12 and 9 wellfields (Figure 9).

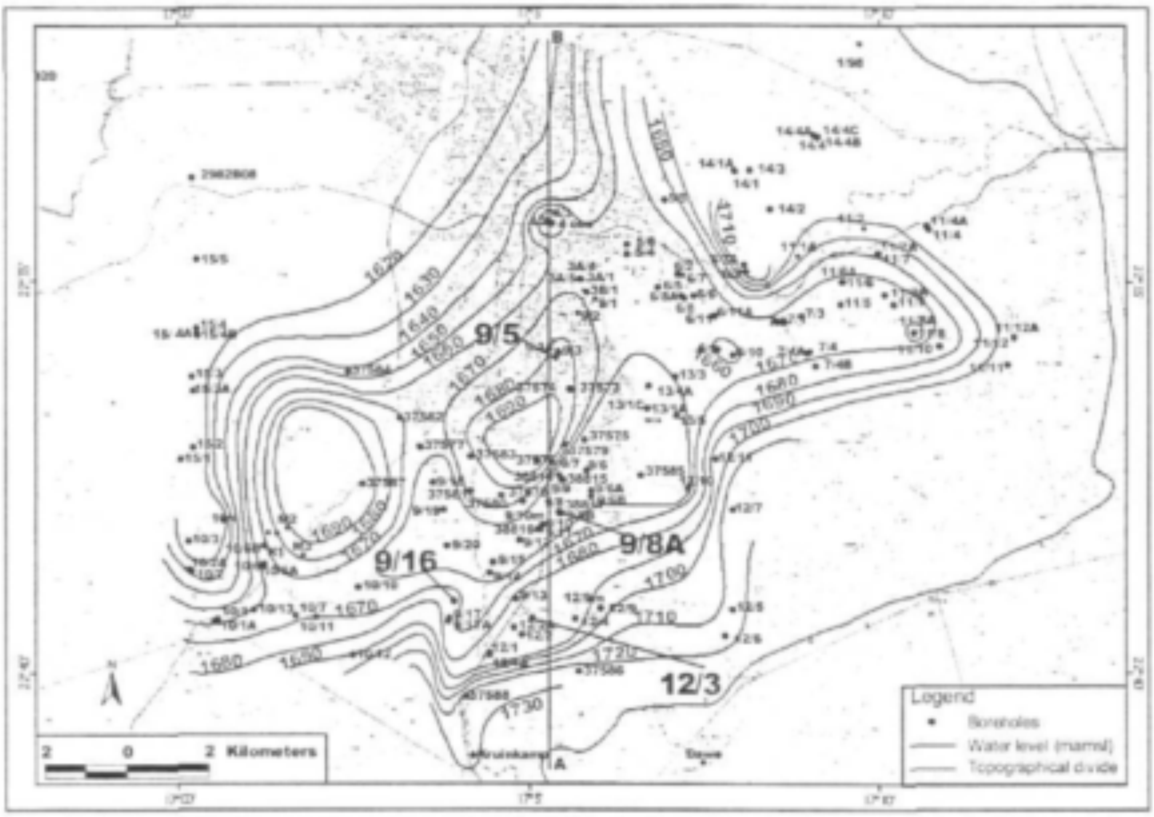


Figure 8 Year 2001 groundwater level contours (Murray and Tredoux, 2001)

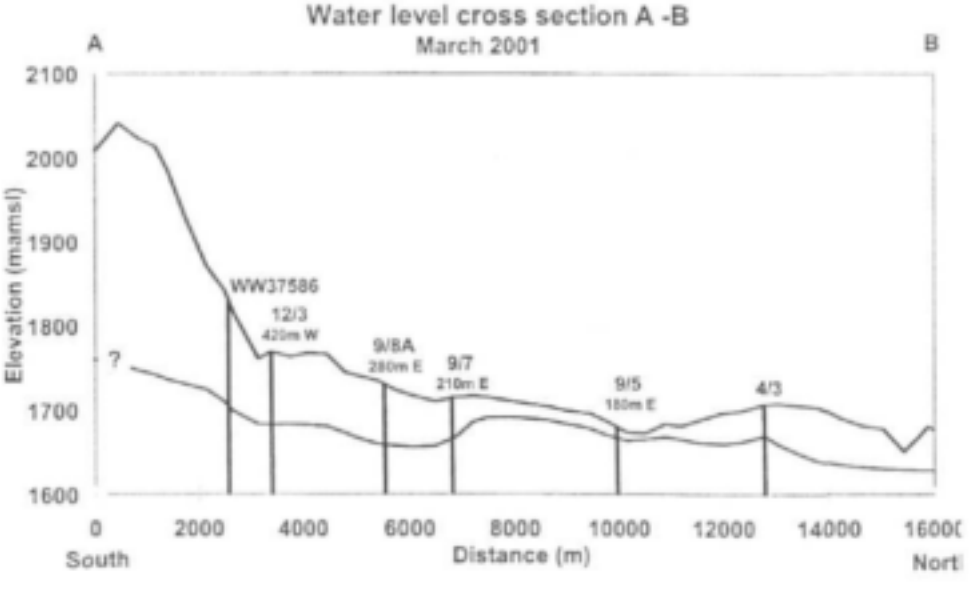


Figure 9 North-south water level cross section, year 2001 (borehole locations are given as distance and direction from the A-B profile line)

3. THE WATER SOURCE FOR ARTIFICIAL RECHARGE

The city's water supply was used as the water source for injection. This fully treated drinking water comes mainly from the dams that supply Windhoek, but a small proportion is derived from groundwater that is pumped from the municipal boreholes.

The water used in borehole injection schemes has to be of a high quality. Important quality criteria for successful injection include very low suspended solids, low concentrations of dissolved organic carbon, the absence of micro-organisms, and acceptable chemical compatibility between the injected water and the natural groundwater (Pyne, 1995, Pérez-Paricio and Carrera, 1999).

In the case of the injection tests, the chemistry of the water is that of a good quality drinking water (Table 3). The microbiological quality of the injected water was also good. A chlorine neutralised sample taken on the10 December 1998 gave a heterotrophic plate count of 2 per 1 mL at 22° C after 72 hours. The internationally recognised target range for drinking water is 0 - 100 counts per 1 mL (Department of Water Affairs and Forestry, 1996). The microbiological sample was taken after the water had been further treated with granular activated carbon and chlorination. The reason for additional treatment with the carbon filter is discussed under borehole 9/8A's injection test.

The chemical compatibility between the injected water and the natural groundwater appears not to be a problem. This concern is related to the potential for chemical precipitation within the aquifer which results in clogging. Because the injection water is lower in dissolved solids (TDS) than the groundwater, particularly with respect to calcium, the injection water dilutes the natural groundwater and, therefore, shifts the calcium carbonate equilibrium, reducing the precipitation potential.

Table 3 Chemical quality of injected water in borehole 9/8A on the 31 July 1998

Determinand	*Concentration*	*Determinand*	*Concentration*
Potassium as K mg/L	8.1	pH (Lab)	8.3
Sodium as Na mg/L	9.5	Saturation pH (pHs) (20deg C)	8
Calcium as Ca mg/L	32	Aluminium as Al mg/L	0.08
Magnesium as Mg mg/L	8.5	Cadmium as Cd mg/L	<0.01
Ammonia as N mg/L	<0.1	Copper as Cu mg/L	<0.05
Sulphate as SO4 mg/L	6.2	Iron as Fe mg/L	<0.05
Chloride as Cl mg/L	13.3	Lead as Pb mg/L	<0.05
Alkalinity as CaCO3 mg/L	119	Lithium as Li mg/L	0.007
Nitrate plus nitrite as N mg/L	<0.1	Manganese as Mn mg/L	<0.05
Ortho phosphate as P mg/L	<0.1	Nickel as Ni mg/L	<0.05
Fluoride as F mg/L	0.2	Strontium as Sr mg/L	0.16
Dissolved Organic Carbon mg/L	<0.1	Zinc as Zn mg/L	<0.05
Conductivity mS/m @25 °C	28	Boron as B mg/L	<0.2
Total Dissolved Solids (Calc) mg/L	179	Bromide as Br mg/L	0.26
Hardness as CaCO3 mg/L	115	Silica as Si mg/L	2.9

4. BOREHOLE INJECTION TEST ON BOREHOLE 9/5

In 1996, Carr Barbour & Associates were requested by the City Engineer, Water Services to investigate the feasibility of injecting water into the aquifer via boreholes for the purpose of artificial recharge. An existing, currently disused production borehole, borehole 9/5 was selected for the first pilot test. This borehole, together with the observation boreholes, borehole 9/3 and 9/3A were drilled into a fault zone consisting mainly of mica schists, and to a lesser extent, quartzites.

The choice of this low-yielding borehole (± 1.4 L/s) was largely based on the limited availability of gravity-fed water and because of the two observation boreholes, Bh 9/3 and Bh 9/3A which are within 150 m of borehole 9/5. The limited water supply for injection was due to the low water levels in the supply dams as a result of the 1994-1995 drought. Although in the 1950's the water level in this borehole dropped by 60 m as a result of pumping, at the time of the injection test in January 1997, the water levels in this area of the aquifer were relatively high (Figure 10).

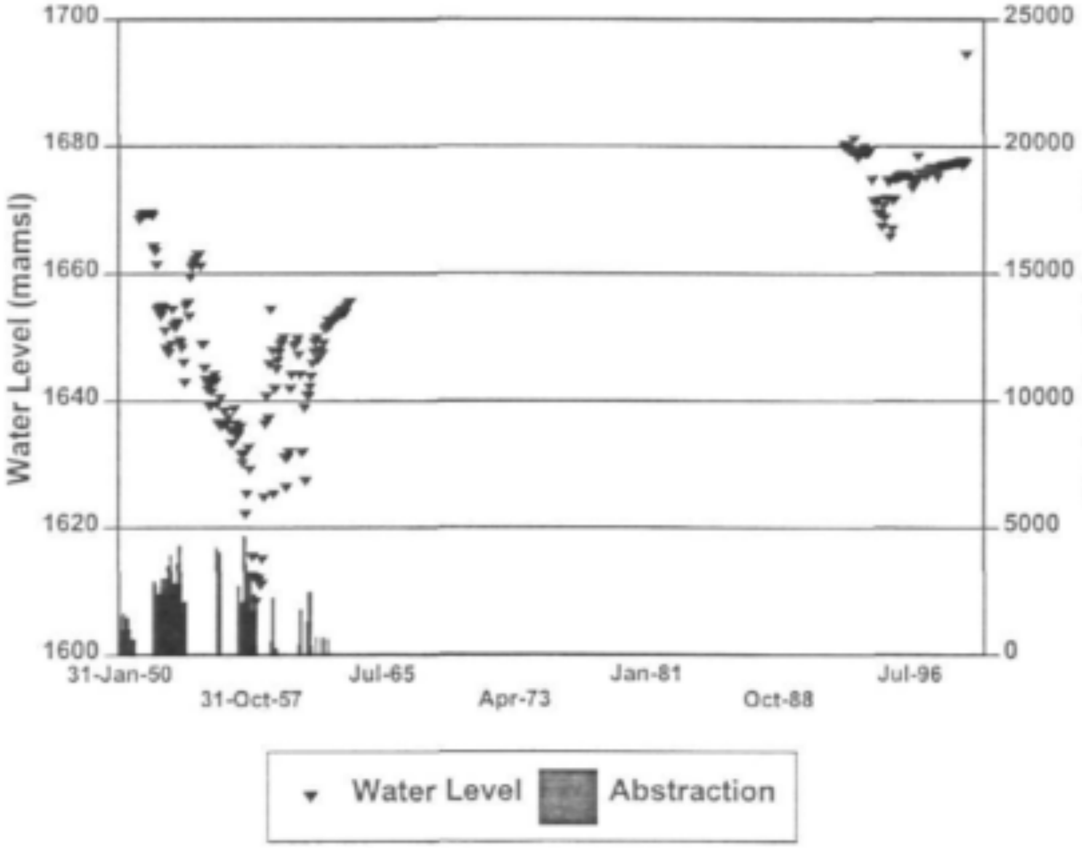


Figure 10 Water level and abstraction data from borehole 9/5

A step injection test was done on the borehole prior to the constant injection test. The step test consisted of six, one hour steps at injection rates ranging from 1.3 L/s to 11.0 L/s. A seventh step at more than 11.1 L/s was attempted, but was stopped due to the rapid rise in the water level and the flow exceeding the capacity of the flow meter.

The constant injection test was run for 21 days between 28 January and 18 February 1997 at an average rate of 2.8 L/s. The water level in borehole 9/5 rose by 6.7 m, and at the end of the test it was 13 m below ground level. After 21 days of recovery, the residual mound was 0.9 m above the original water level prior to the injection test.

In observation boreholes 9/3 and 9/3A the water levels rose by approximately 3.6 m. During the recovery test, the water level readings from the data loggers displayed a sinusoidal pattern that correlate to earth tides. This indicates that the aquifer in this area behaves in a confined manner.

The aquifer parameters obtained from test pumping and the injection test are given in Table 4.

Table 4 Aquifer parameters obtained form pumping and injection (Carr Barbour & Associates, 1997)

Borehole	*Transmissivity (m^2/day)*		*Storativity*	
Number	*Pump Test*	*Injection Test*	*Pump Test*	*Injection Test*
9/5	13	18	0.034	-
9/3	-	13	-	0.034
9/3A	10	14	-	0.032

Following this test, Carr Barbour & Associates recommended that more long-term tests should be undertaken (CBA, 1997).

5. BOREHOLE INJECTION TEST ON BOREHOLE 9/8A

This borehole was drilled into a permeable fault zone. The geological logs show that the borehole penetrates laminated, fractured, sericitic quartzites from 5 m to 138 m; graphitic schist from 138 m to 165 m; and intercalated fractured, sericitic quartzites and graphitic schists from 165 m to its completion depth of 202 m. The drilling chippings are fracture stained throughout the depth of the borehole, although samples from the last 20 m to 30 m of the borehole appear to be less fracture stained. The borehole was drilled in 1994 near borehole 9/8 which had collapsed. Plates 1 and 2 show the borehole at the time of the injection test, with a carbon column standing next to the borehole.

5.1 Suitability for injection

Water strikes were recorded at 100 to 108 m; 115 to 117 m and 168 to 173 m below surface. It was not possible to measure the drilling (or "blow") yields of the water strikes because of the vast quantity of foam that was required in the drilling process. The rest water level prior to test pumping in 1995 was 88.79 m below surface. When borehole 9/8 was drilled in 1958, the water level was 40.8 m below surface. Over the past forty odd years, abstraction has led to a drop in the water level of nearly 50 m (Figure 11).

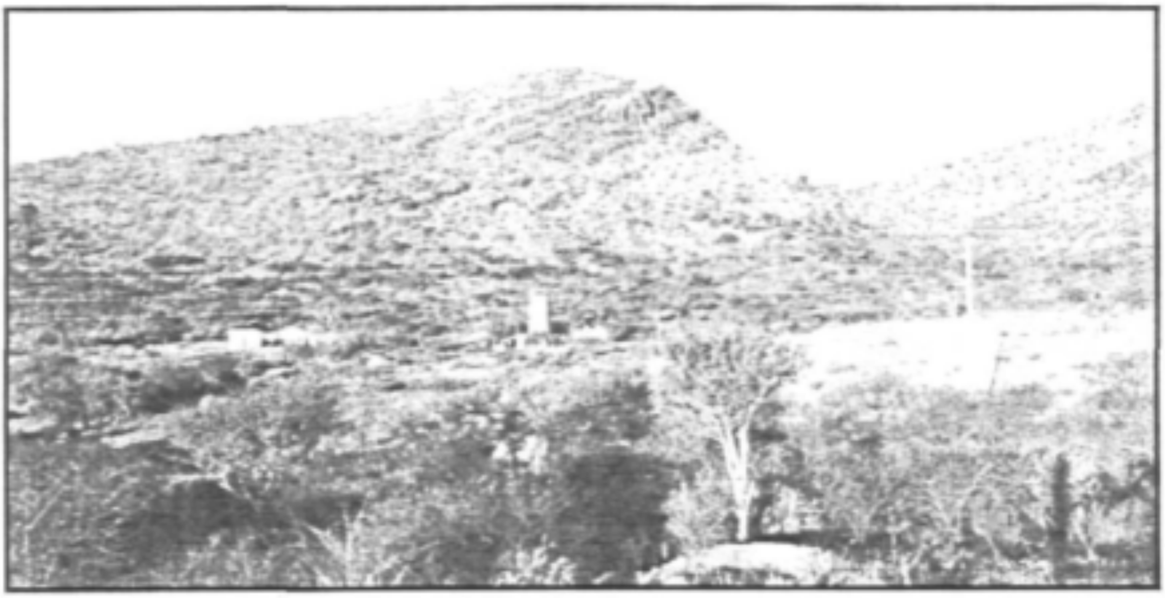
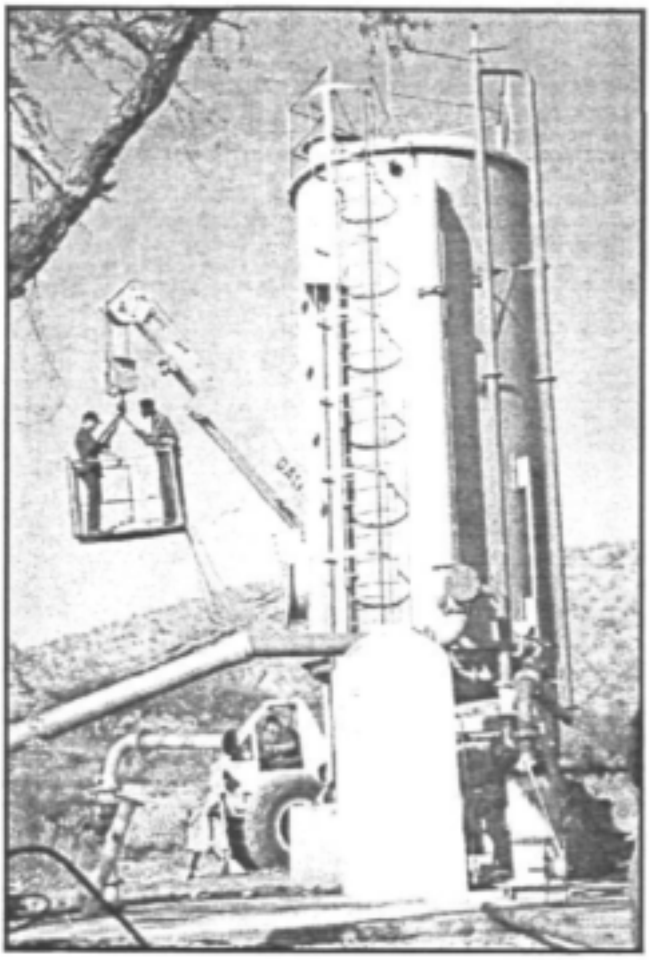


Plate 1 Borehole 9/8A in the centre of the picture with the carbon filter standing upright. The Kleine Kuppe Formation micaceous quartzites can be seen outcropping in the background.

Plate 2

Borehole 9/8A: Erecting the carbon filter. The borehole is to the right of the white box which houses the chlorinator.



Before the start of the injection test, this part of the aquifer had been heavily pumped and was characterised by a hydraulic cone of depression (Figures 12; 13 and 14). The long-term water level graph (Figure 11) indicates that it would take more than five years without significant pumping for the water level to "recover", and that full recovery to the original, pre-pumping water levels, would take much longer. Thus, as far as available storage space is concerned, this site was suitable for artificial recharge in 1998.

This part of the aquifer also appeared suitable for receiving water in terms of it's permeability. The three day constant discharge test done on borehole 9/8A in 1995 at a rate of 12.5 L/s, gave a transmissivity value of 60 m^2/day, based on the saturated thickness at that stage. Given that the borehole logs show signs of intense fracturing above the 1998 water level, and that water strikes were encountered below the 1998 water level, this borehole was seen as a favourable injection site.

Unfortunately, for injection purposes, the borehole was cased from surface to 112.9 m with plain steel casing (273 mm OD), and thereafter, to it's completion depth at 202 m, with slotted steel casing.

The injection pipe was installed to 150 m below surface. Water entering the borehole at this point could flow into the aquifer at the following depths:

1. the water strike zones around 115 to 117 m and 168 to 173 m (where there is slotted casing);
2. it could rise up the annulus (between the plain casing and the rock) to enter the water strike zone around 100 to 108 m; and
3. it could rise up the annulus to enter permeable areas within the unsaturated zone above the water level (that is, above 84 m).

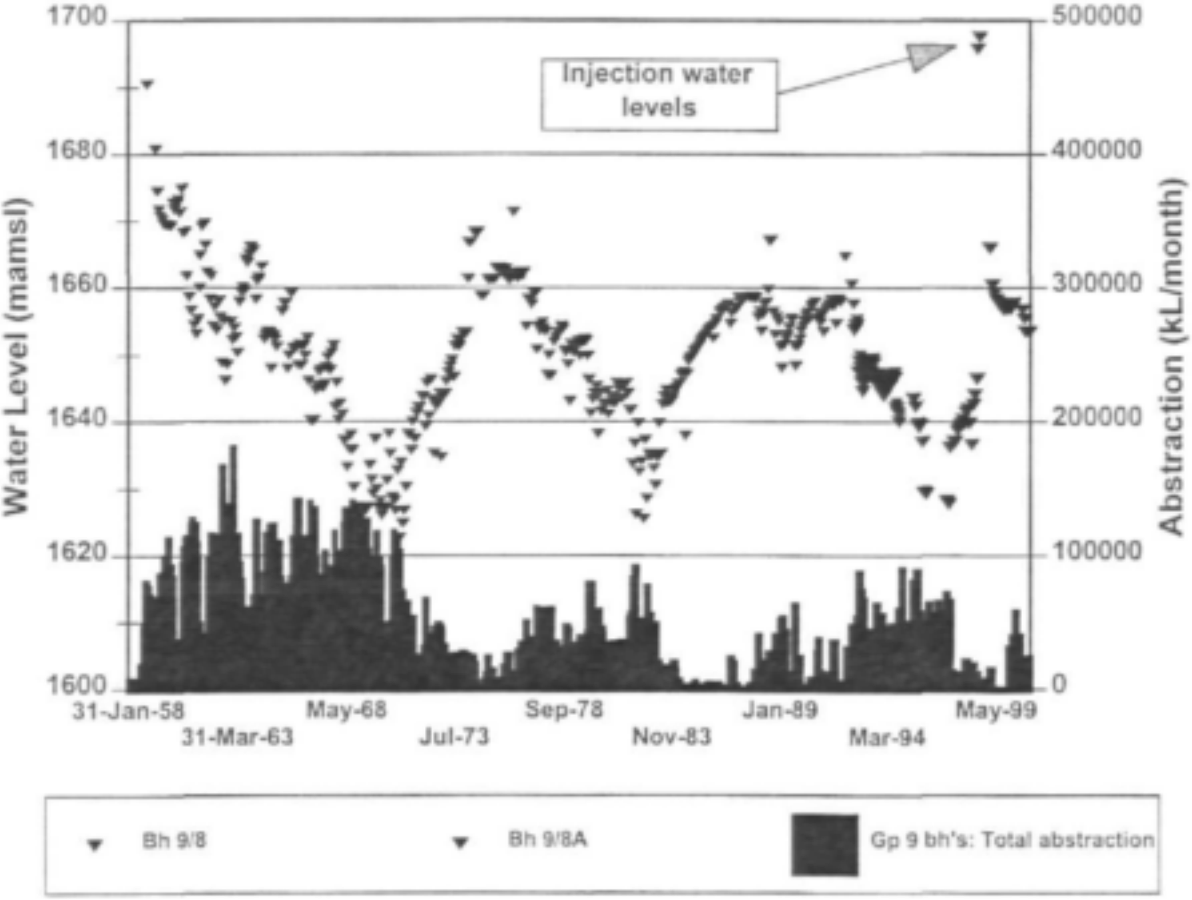


Figure 11 Water levels and abstraction data from boreholes 9/8 and 9/8A (the drop in the water level during the early 1990's was due to abstraction from nearby boreholes)

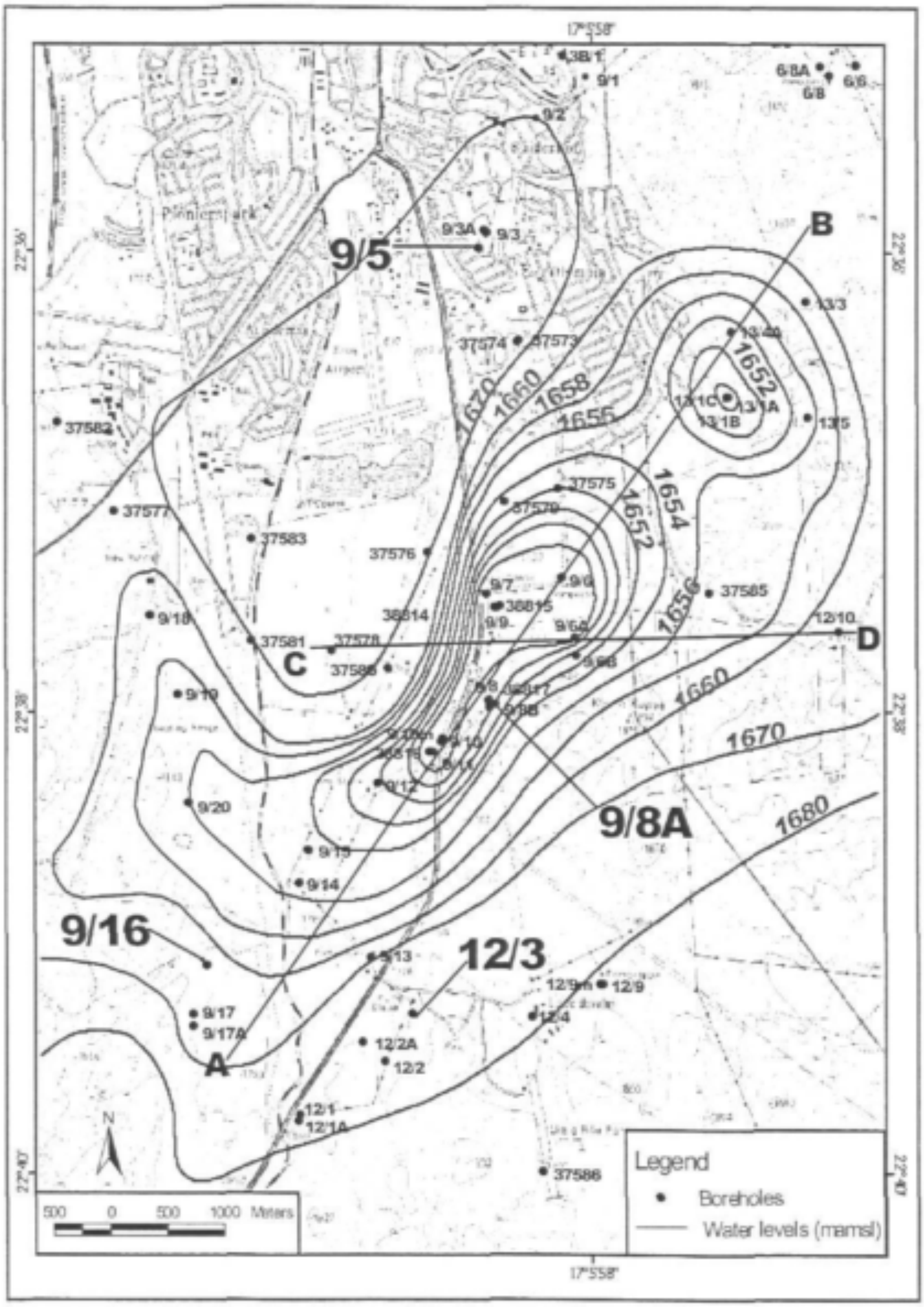


Figure 12 Water levels contours in August 1998 prior to borehole injection

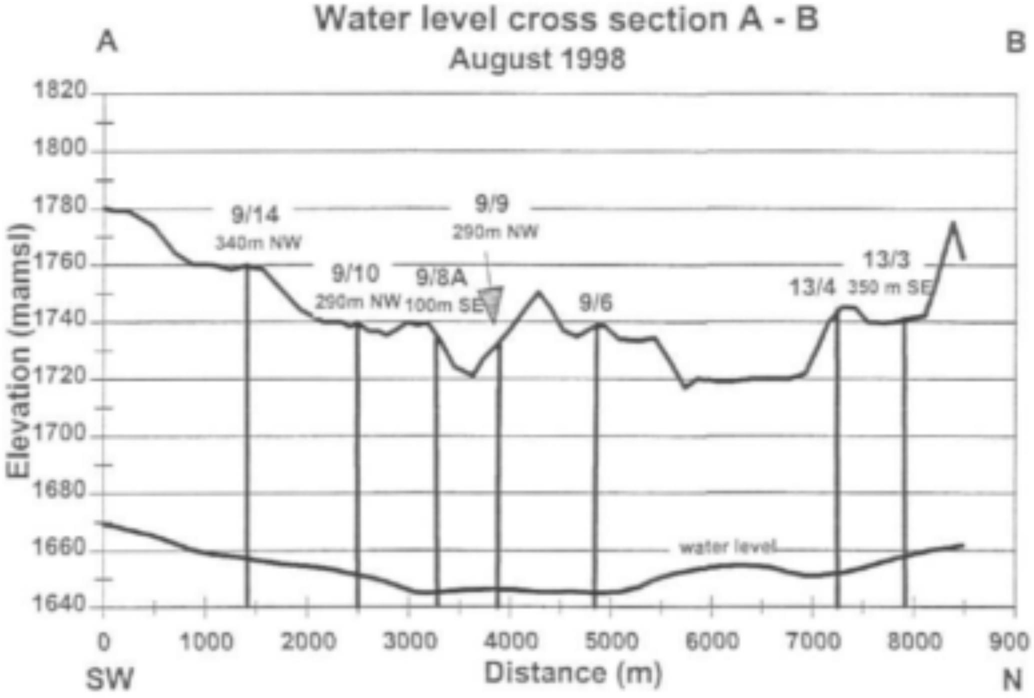


Figure 13 SW-NE water level profile in August 1998 prior to borehole injection (borehole locations are given as distance and direction from the A-B profile line)

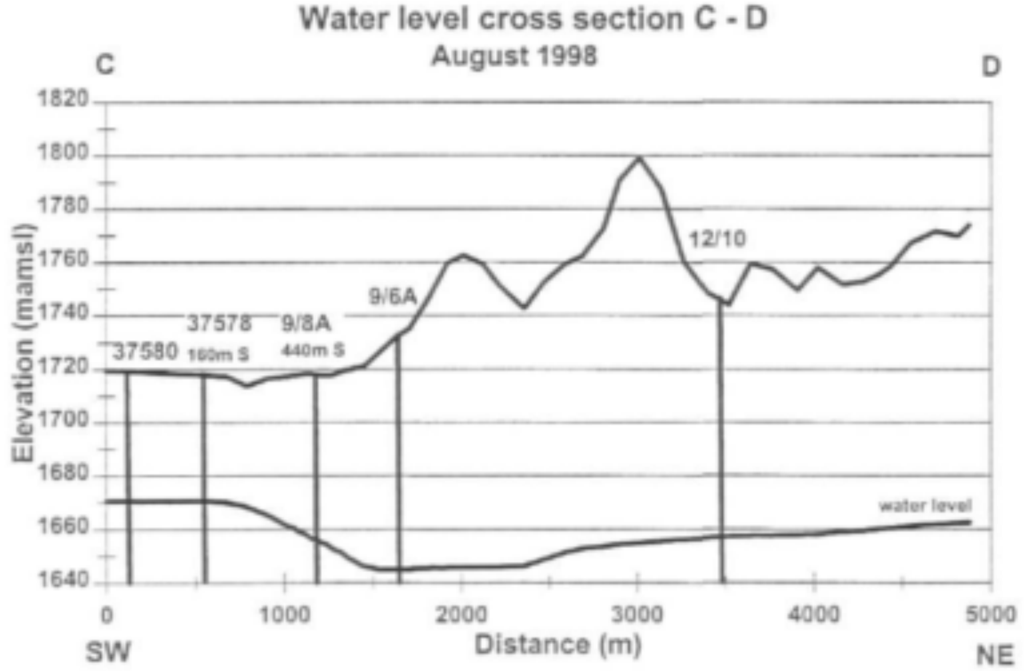


Figure 14 West-east water level profile in August 1998 prior to borehole injection (borehole locations are given as distance and direction from the C-D profile line)

5.2 *The injection test*

A **step injection test** was done on borehole 9/8A to establish a suitable rate for the constant injection test. The results of the step test are given in Figure 15. The water level drop during the third step is probably a result of the initial inefficiency of the borehole to allow water to enter the aquifer via the annular column. In order for water to overcome this "blockage" it rose within the borehole until the pressure was sufficient to force the water past the "blockage" and into a permeable fracture zone. Once water could easily enter this fracture zone, the water level in the borehole stabilised until the fracture zone could not receive water at a high rate any more. This may account for the flattening of the curve after 20 minutes into the 3rd step until 40 minutes into the 4th step.

Interestingly, the borehole chippings (taken during drilling) show a fairly high degree of fracture staining at this level. Large, fracture stained chippings of up to 20 mm in length, were obtained at depths 65 m to 70 m below surface which correlate to the flattening of the curve at a depth of 64 m to 66 m below surface (or 20 m to 22 m above rest water level in Figure 15). This shows that during injection, a certain portion of the water enters the unsaturated zone.

It was decided to do the constant injection test at the maximum capacity of the carbon column. This started at 21.4 L/s and dropped with time to an average of 17.1 L/s.

The recovery test (Figure 16) that followed the step injection test shows the presence of a fracture zone at 75 m below surface. This also corresponds to a zone of fracturing at 75 m which was identified from the drilling chippings. The recovery data does not, however, capture the apparent fracture at 65 m which was highlighted in the step injection test.

The **constant injection test** ran for 195 days, and the total volume of water that was injected was 288 617 m^3. The average injection rate was 17.1 L/s. Figure 17 shows the water level rise in this borehole. The injection was interrupted four times for back-flushing the carbon filter. The last stoppage reflects the water level decline on completion of the 195 day injection test.

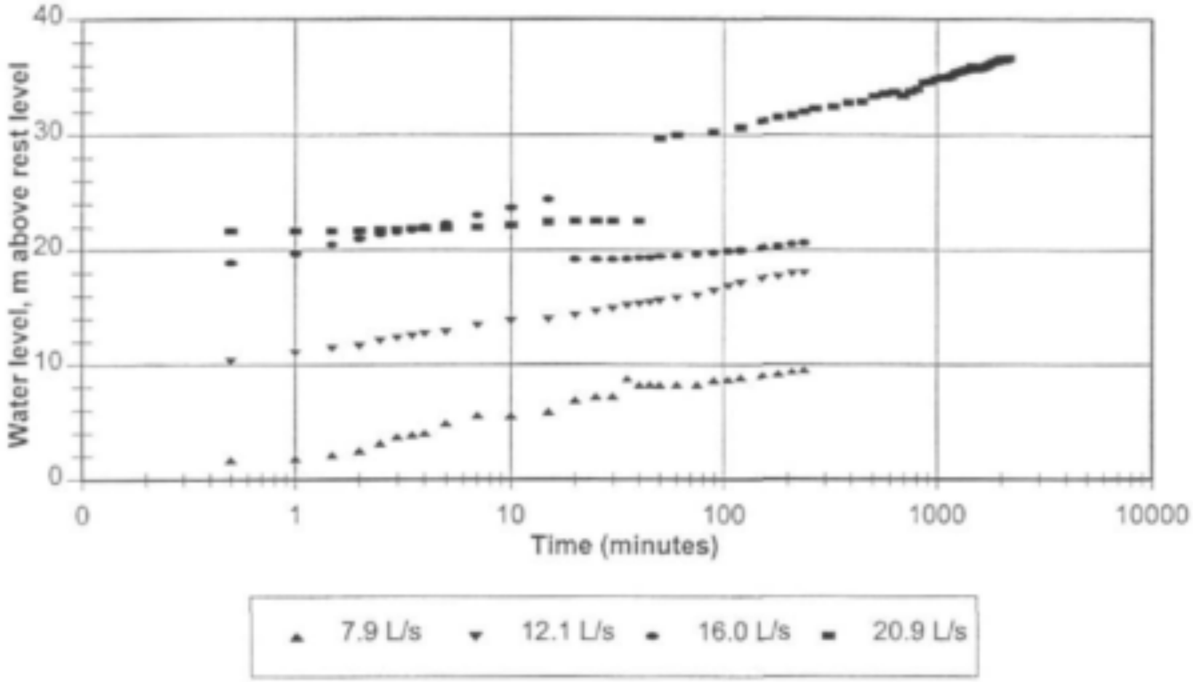


Figure 15 Borehole 9/8A: Step Injection Test (Rest Water Level = 85.93 m below surface)

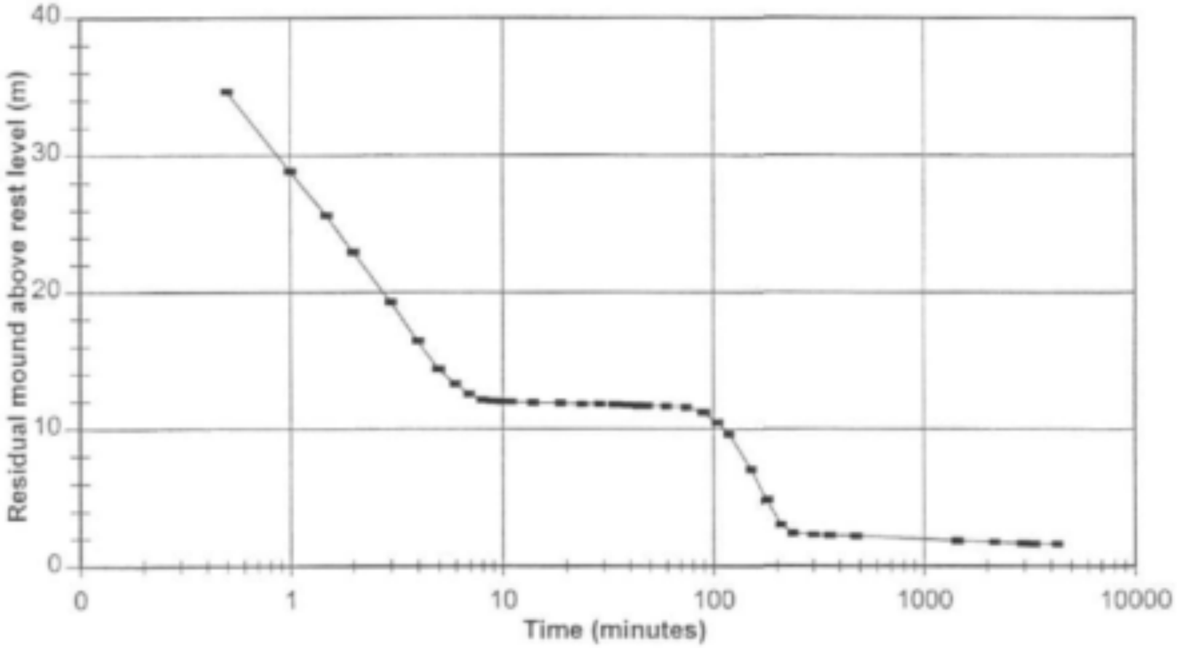


Figure 16 Artificial recharge mound dissipation following the step injection test

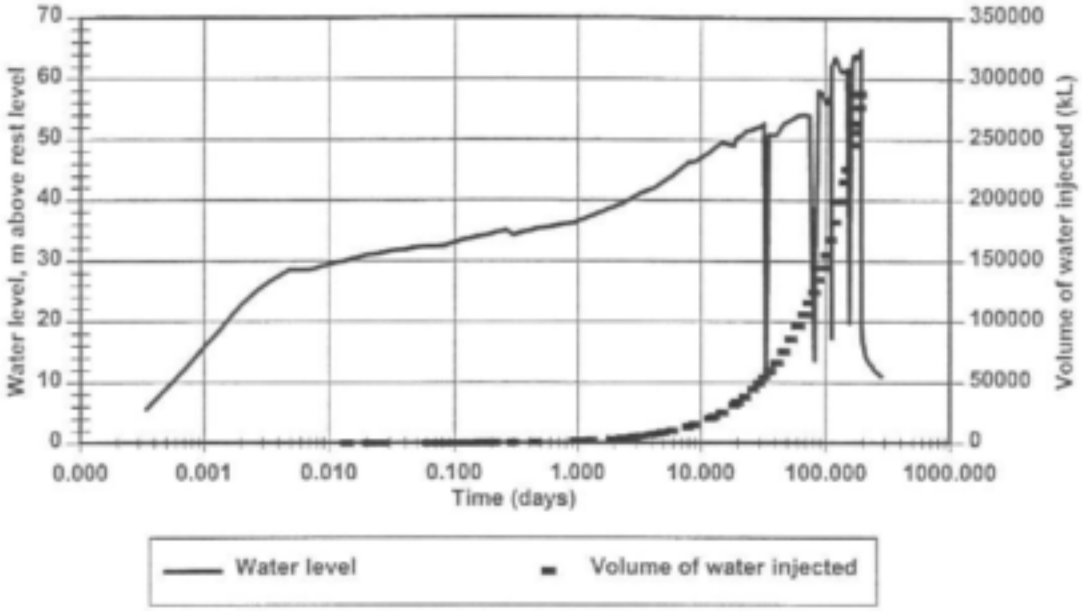


Figure 17 Borehole 9/8A: Constant Injection Test (Starting Water Level = 84.04 m below surface; Average injection rate = 17.1 L/s)

5.3 *The regional effect of injection*

Figures 18, 19 and 20 show the water level response in boreholes around borehole 9/8A. These figures show that the response is remarkably quick. Borehole 9/12, for example, is over a kilometre from the injection borehole and it appears to have responded virtually immediately. This indicates that the aquifer is at least partly confined. This is confirmed by water level readings from the data loggers which display a sinusoidal pattern that correlate to earth tides. However, geological logs of boreholes which are located within fault zones show fracture staining on the rock chippings from the surface onwards, indicating that in certain areas of the aquifer, the conditions are not necessarily confined.

The reason for the dramatic water level response in boreholes 9/10 and 9/11 (Figure 18) is believed to be a result of the lower permeability in the direction of these boreholes (as confirmed by the lower transmissivity values obtained during pumping and injection tests, and the lower throughflow observed during dilution tests). It could also be due to a lower storativity in that part of the aquifer, or a combination of both, however, the pumping and injection test data indicates that it is more likely to be a result of lower permeability than storage.

The injection test also highlighted that the aquifer is compartmentalised. Boreholes 9/6A and 9/6B are 139 m apart, yet their water levels prior to injection differed by nearly 10 m (Figure 20). During injection, borehole 9/6A's water level rose at the same rate as that of borehole 9/6, but borehole 9/6B barely responded. After injection their water level differences were far less than prior to injection.

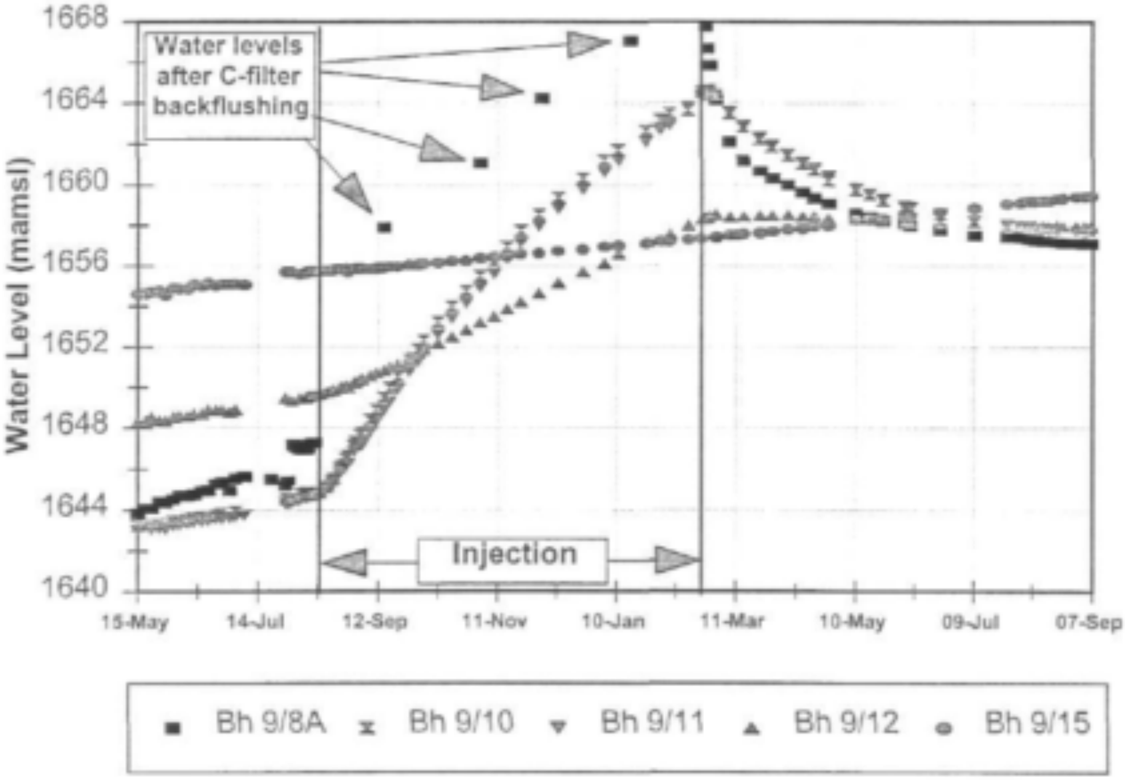


Figure 18 Water level response to injection in boreholes south-west of borehole 9/8A

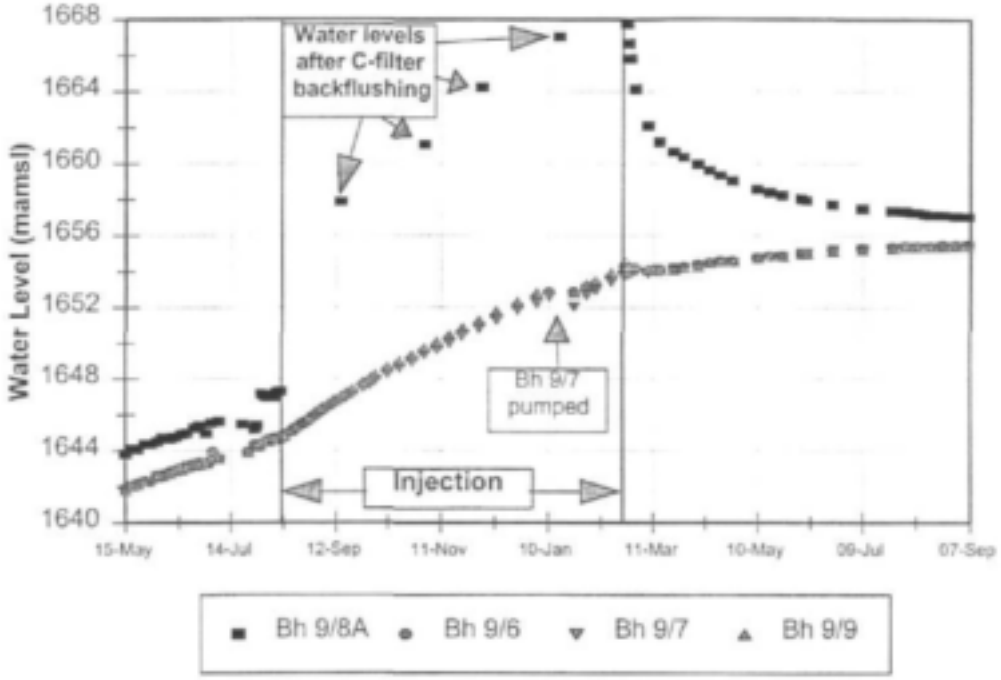


Figure 19 Water level response to injection in boreholes north of borehole 9/8A

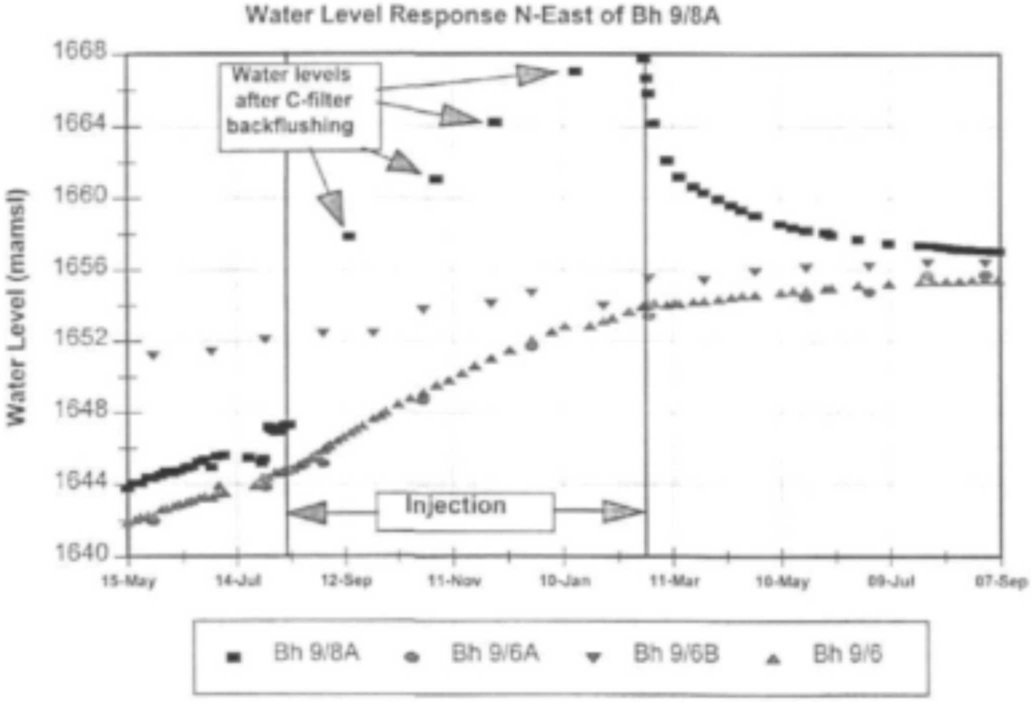

Water Level Response N-East of Bh 9/8A

Figure 20 Water level response to injection in boreholes east of borehole 9/8A

Figure 21 presents contours of the water level rise after injection. The values were obtained by determining the natural water level rise that would have taken place without injection, and subtracting this from the measured water levels. Because good records were kept during the pre-injection period, the natural water level rise could be accurately estimated.

Although the injection of 289 000 m^3 had a significant local effect around borehole 9/8A, it is clear that a far greater volume of water would need to be injected into this part of the aquifer in order to have a considerable effect on the water levels in the Group 9 and 13 parts of the aquifer.

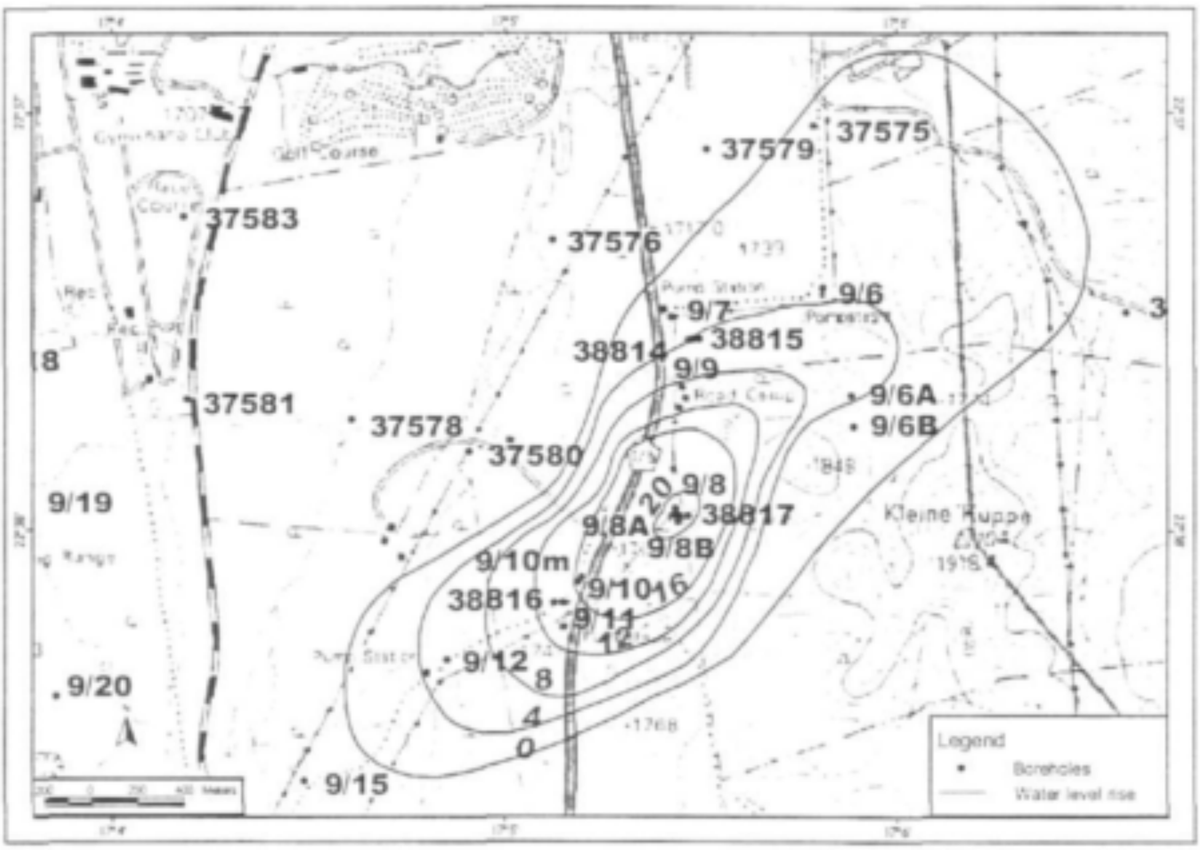


Figure 21 The effect of injection in borehole 9/8A on the aquifer

5.4 Aquifer permeability and storage

The injection test on borehole 9/8A provided the opportunity to reassess the aquifer parameters. This was done by conducting dilution and tracer tests, re-evaluating the pumping test data, evaluating the injection data in a similar manner to standard pumping test analyses, and by drawing water level rise contours in order to obtain a storativity value.

Dilution tests were carried out on boreholes 9/11 and 9/9 for determining the flow through the boreholes. These tests indicated a high flow through borehole 9/9, and low flow through borehole 9/11. The zone of high flow in borehole 9/9 was traced to depths 95 m to105 m below surface. A tracer test, using a fluorescent dye, was done with the assistance of the Institute for Groundwater Studies between boreholes 9/8A and 9/9 (Figure 22). A remarkably high flow velocity of at least 3.6 m/min was obtained between these boreholes which are located 786.5 m apart (Van Wyk, Murray and Van Tonder, 2000). This showed that the injection water flowed at a rapid rate along the highly permeabile fault zones on which these boreholes were sited.

Previous pumping test analyses on borehole 9/9 gave a transmissivity value of 294 m^2/day from the recovery data (Colvin, 1994). Review of this data indicates that the fracture transmissivity could be as high as 4 000 m^2/day and the formation transmissivity about 200 m^2/day. Injection test curves from the boreholes north of 9/8A (ie 9/6; 9/7 and 9/9) give an average fracture T-value of 1 600 m^2/day and a formation T-value of 160 m^2/day. The average S-value obtained from the injection curves north of borehole 9/8A was 0.004.

South-west of the injection borehole, the permeability of the aquifer seems to be lower. Pumping test T-values for boreholes 9/10 and 9/11 yield the following values: Fracture transmissivity, 120 -270 m^2/day; and formation transmissivity, 35 m^2/day. The S-value obtained from pumping test data of boreholes 9/11 is 0.008. The injection curves gave similar formation T-values and S-values, but higher fracture T-values. The average fracture T-value is 900 m^2/day; the average formation T-value is 45 m^2/day; and the average S-value is 0.004.

Possibly a more realistic method of determining the S-value is by using the water level contours in Figure 21 to compare the volume of rock that was saturated during the injection process with injected volume. The volume of water injected into borehole 9/8A was 0.289 Mm^3 and the volume of rock saturated is estimated from Figure 21 to be 24.4 Mm^3.

The S-value obtained was:

S = volume injected /saturated rock volume

= 0.289 Mm^3 / 24.4 Mm^3
= 0.012

This value is considerably higher than the 0.004 value obtained from analysing the injection curves. Error could be introduced by not considering natural recharge during the injection period, or by incorrectly estimating the saturated volume of rock.

Rainfall during the injection test was 126.4 mm. This is nearly half the average (222.8 mm) for the same period. Considering that the area under consideration is mostly covered by soil and to some extent by road and housing developments, it is unlikely that natural recharge (which is unlikely to exceed 2 % of rainfall) had much influence on groundwater levels.

Two options exist in underestimating the saturated rock volume. Either the injected water had an effect on the Group 13 area and thus the area of influence extended further towards the north-east than initially recognised, or injected water occupied available storage within the micaceous quartzites beneath the schists. The S-value obtained by re-drawing the water level rise contours to accommodate these options are 0.010 and 0.008 respectively.

Extending the contours to include the Group 13 wellfield did not affect the calculated S-value, however, the value dropped by 0.003 after assuming that water could occupy space beneath the schists. This scenario is based on the assumption that the two water levels exist at one point in certain parts of the Group 9 area. The one water level is associated with the schists, and is the commonly measured water level, and the other is associated with the micaceous quartzites below the schists, and would only be measured in a borehole that seals off the shallow, schist water level. Should this scenario exist, injected water could feasibly flow under the schists to occupy available space within the micaceous quartzites. Although this option yields a lower S-value, there is no evidence that two water levels do exist where the quartzites underlie the schists.

The same approach to obtain a S-value was used with abstraction data. The high abstraction period between 1992 and 1996 was used, and this gave a S-value of 0.008 (Murray, in prep). This value is possibly the most realistic, since the volume of water pumped and the volume of rock mass dewatered was far greater than the volumes used with the injection data.

The manner in which the aquifer responds to abstraction and injection, and the storage coefficients obtained from this data suggest that the aquifer is generally semi-confined. In this highly anisotropic aquifer, the transmissivity and hydraulic conductivity values vary considerably. The S-value remains difficult to determine with a high degree of certainty. The values obtained from the injection data range between 0.004 and 0.01. The actual value is estimated, at this stage, to be about 0.008, which is the value that was obtained after four years of excessive pumping. This value is likely to represents the storativity of the quartzites; the value for the schists is likely to be significantly lower.

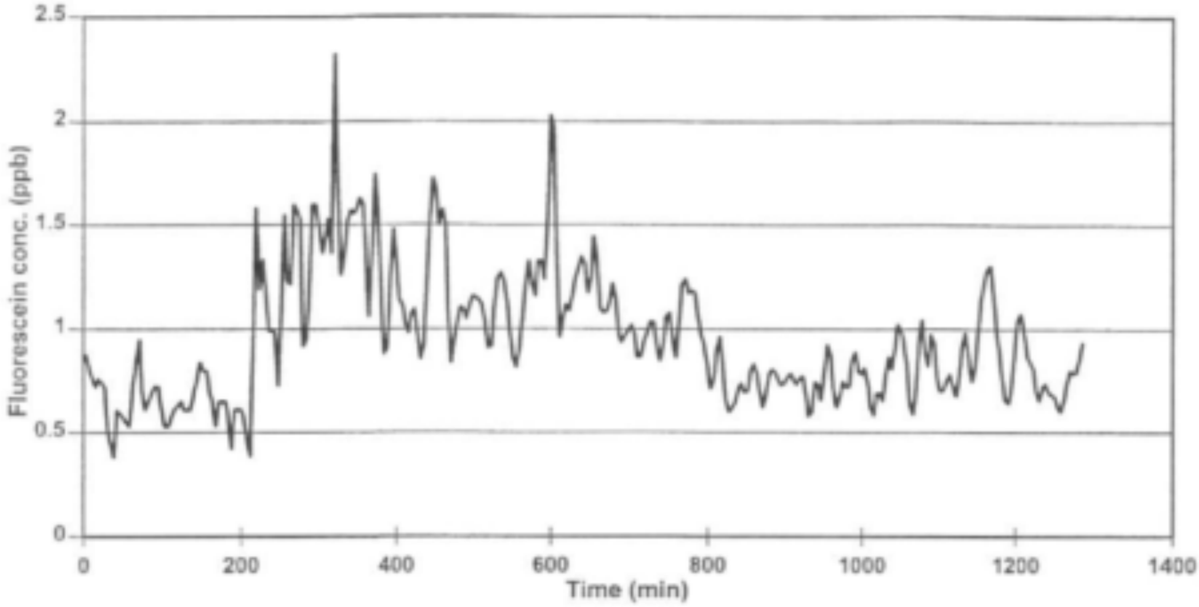


Figure 22 Results of the tracer test between boreholes 9/8A and 9/9, 786.5 m apart

5.5 *Water quality issues*

The water source was good quality drinking water from the town supply system. In the case of the injection test on borehole 9/8A, the water was further treated by percolation through a granular activated carbon filter, followed by chlorination. As a result, the injected water was of a very high quality.

Percolation of the water through the carbon filter mechanically screened the water, removing all remaining suspended material, which ensured both a very low turbidity (<< 1 NTU) and a low organic carbon content of approximately 3 mg/L DOC (Figure 23). Chlorination of the injected water (to 0.3 mg/L free chlorine), after passing through the carbon filter, ensured disinfection.

The reversal of DOC values before- and after-carbon filtration, shortly before the second backflush, could either be an analytical error, or it could indicate that the carbon filter lost its effectiveness after a long operational period.

The chemical compatibility between the injected water and the natural groundwater should not be a problem, since the injection water is lower in dissolved solids than the groundwater (Table 5).

An interesting phenomenon happened with regard to water quality during the injection test. Between the 5th and 26th October 1999, the water quality in borehole 9/11 changed significantly. The concentrations of sodium, magnesium, calcium, chloride and sulphate rose dramatically, and then, by early December dropped back to their original levels. Figures 24 and 25 show this rapid rise in electrical conductivity and the sulphates.

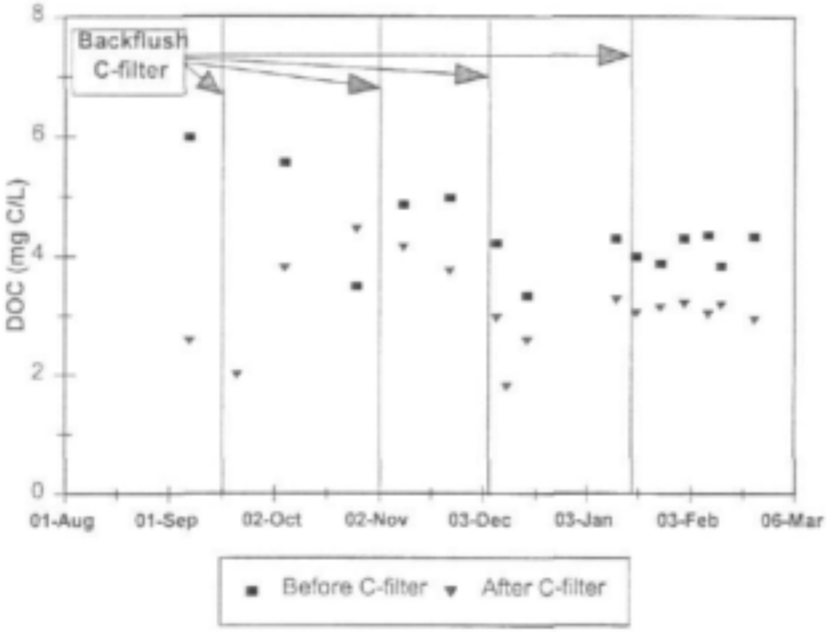


Figure 23 Dissolved organic carbon levels of injected water into borehole 9/8A before and after treatment with granular activated carbon

It is postulated that sodium, calcium and magnesium sulphates were present in the upper, dewatered portion of the aquifer. The water level in the vicinity of borehole 9/11 increased due to the injection of water in borehole 9/8A, and these salts were dissolved and caused the increase in salinity noted at borehole 9/11. Gradually these salts were leached and diluted as the water slowly migrated past the borehole. For this reason an initial high peak appeared which gradually dissipated.

The source of these sulphates may possibly be ascribed to sulphide minerals, particularly pyrite, which is known to occur in fractures in the quartzites. Over the past forty two years, since pumping dropped the water levels in this area, the vadose zone has become well aerated and the sulphides have been oxidised, and have formed sulphates. Once the groundwater level rose it dissolved the sulphates and associated ions, and hence the observed peak. Of interest though, is that the alkalinity and pH did not drop at the same time as the rise in sulphates, as one would generally expect. It may, however, be that sufficient calcareous material is available for re-establishing chemical equilibria.

Table 5 Comparison between injection and groundwater quality - July 1998 (the pronounced differences are highlighted)

Determinand	*Bh 9/8A*	*Injection Water*
Potassium as K mg/L	7.4	8.1
Sodium as Na mg/L	**36**	**9.5**
Calcium as Ca mg/L	**78**	**32**
Magnesium as Mg mg/L	**33**	**8.5**
Ammonia as N mg/L	<0.1	<0.1
Sulphate as SO4 mg/L	**120**	**6.2**
Chloride as Cl mg/L	7.8	13.3
Alkalinity as CaCO3 mg/L	**283**	**119**
Nitrate plus nitrite as N mg/L	<0.1	<0.1
Ortho phosphate as P mg/L	<0.1	<0.1
Fluoride as F mg/L	0.3	0.2
Dissolved Organic Carbon mg/L	<0.1	<0.1
Conductivity mS/m @25 °C	**74**	**28**
pH (Lab)	7.7	8.3
Saturation pH (pHs) (20deg C)	7.3	8.0
Total Dissolved Solids (Calc) mg/L	**474**	**179**
Hardness as CaCO3 mg/L	**331**	**115**

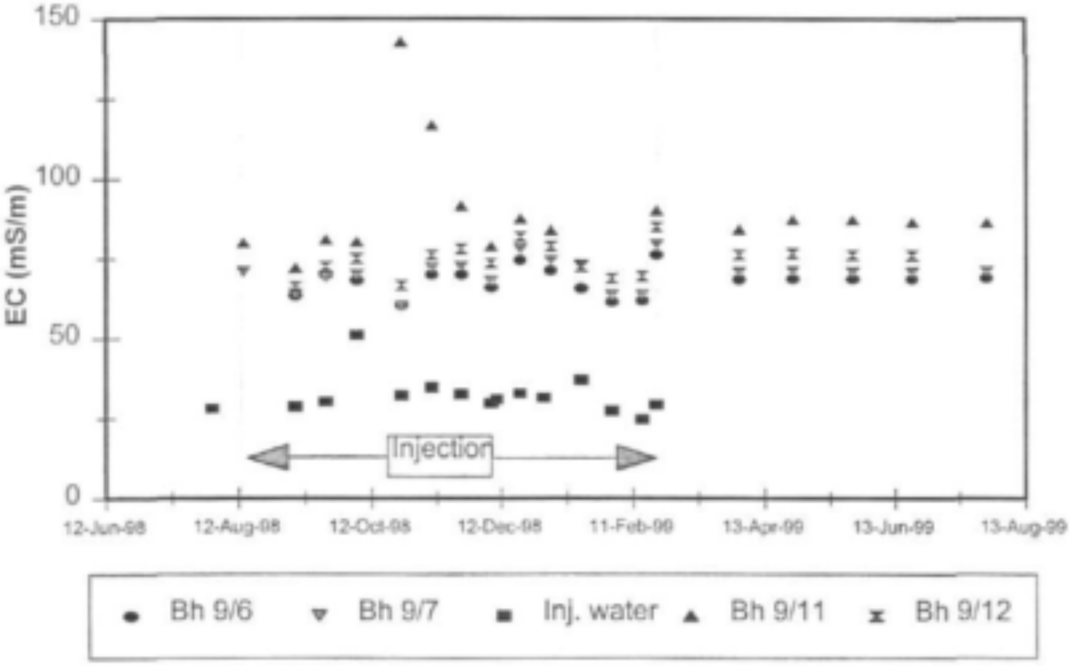


Figure 24 The sudden rise in electrical conductivity at borehole 9/11

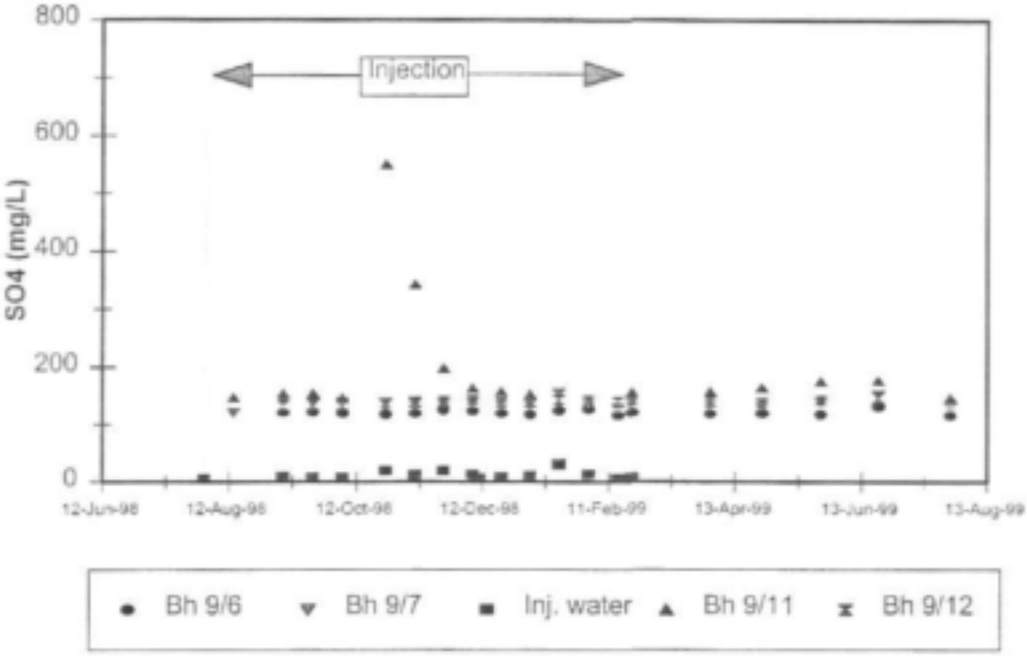


Figure 25 The sudden rise in the sulphate concentration at borehole 9/11

5.6 *Conclusions from borehole 9/8A injection test*

The injection test into borehole 9/8A showed that this part of the Windhoek aquifer is receptive to artificial recharge. The tracer test demonstrated that the injected water flows rapidly along fractures at rates exceeding 3.6 m/min. From pumping and injection test analyses, it appears as if the fracture transmissivity can be an order of magnitude greater than the formation transmissivity. While fracture T-values in excess of 1000 m^2/day are not unrealistic, the formation T-values obtained were in the order of 180 m^2/day to the north of borehole 9/8A, and 40 m^2/day towards the south-west of borehole 9/8A.

The storativity values obtained from pumping tests, the injection test and historical water level data range between 0.004 and 0.01. The value of 0.008 that was obtained from historical water level data seems to be the most realistic at this stage.

The injection test showed that water levels respond rapidly to injection. This even holds for some boreholes over a kilometre from the injection borehole. This rapid response, together with the sinusoidal water level fluctuations that reflect earth tides, indicate that the aquifer is at least partly confined.

The long term objective of recharging the aquifer in the region of borehole 9/8A would be to rapidly fill the hydraulic depression that is created in this area as a result of pumping. This depression was most pronounced at the end of the 1960's after a decade of intensive pumping where 10 Mm^3 was drawn from the Group 9 boreholes. Borehole 9/8A's injection test demonstrated that this depression can be rapidly re-filled. Assuming four injection boreholes are used with similar capacities to that of borehole 9/8A, then approximately 2 Mm^3/a could be injected into this part of the aquifer.

6. BOREHOLE INJECTION TEST ON BOREHOLE 9/16

This borehole was drilled into a permeable fault zone linked to the western part of the Windhoek graben (Plate 3). Figure 5 shows that the water level at this part of the Windhoek aquifer is about 12 m higher than the water level within the cone of depression around borehole 9/8A, and therefore injected water will flow towards the abstraction boreholes located in the centre of the depression.

The borehole 9/16 - 9/17 area (and even further to the south) is thought to be an ideal site for injection for three reasons: Firstly the Venusberg fault runs through this area, and therefore high injection rates should be achieved because of the high hydraulic conductivities associated with large fault systems; secondly, by injecting water as far south as possible, it will have the effect of replenishing what is believed to be the main storage area of the aquifer, namely, the Auas quartzites; and thirdly, by injecting up-gradient of the Group 9 production boreholes, it is unlikely that artificially recharged water could be lost from the aquifer.

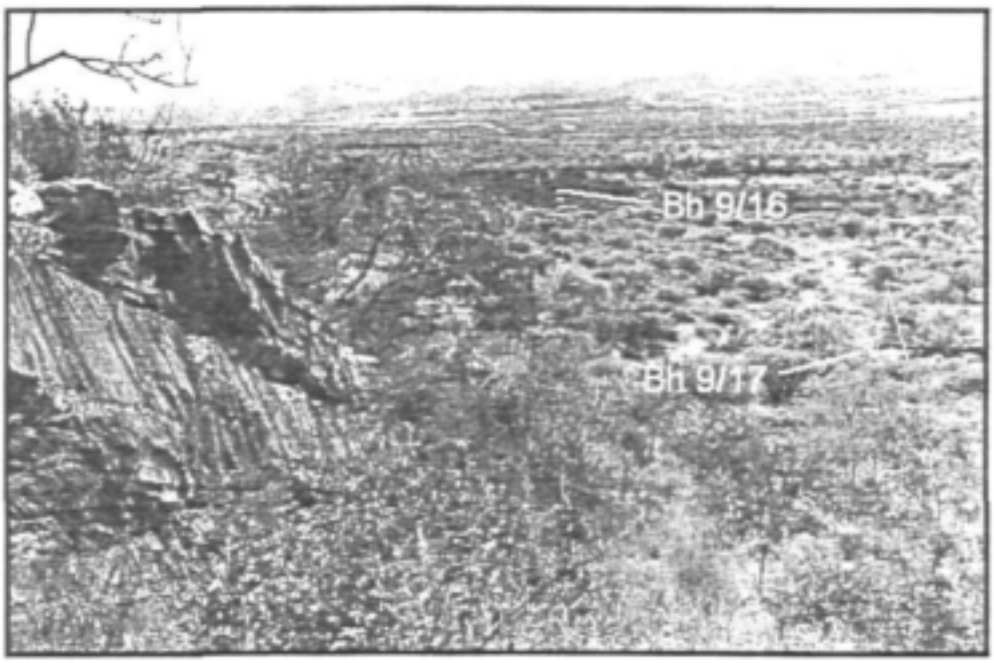


Plate 3 Fault on which borehole 9/17 was sited. Borehole 9/17's tripod is on the right hand side of the photo and borehole 9/16's tripod is in the upper centre part of the photo.

6.1 Suitability for injection

The borehole logs are not available, however it is known that the main water strike was encountered at about 120 m below surface. The borehole is cased from surface to its completion depth of 127 m, which, as in the case of borehole 9/8A, is not suitable by design for injection. The drilling yield was reported as being 15 L/s. The late-time transmissivity value obtained from a three day constant discharge test at 8.8 L/s was 43 m^2/day. This reflects the flow from the matrix or micro-fissures to the main fractured part of the aquifer.

It must be noted that boreholes in this part of the aquifer generally do not penetrate the entire thickness of the fractured quartzites, and therefore aquifer parameters obtained from discharge-drawdown relationships may be incorrect. Borehole 9/17A, drilled 500 m from borehole 9/16 and also on the western edge of the graben structure, had to be abandoned during drilling due to continual collapse as a result of the intensity of the fracturing. This suggests a very high permeability in this part of the aquifer. Like borehole 9/17A, borehole 9/16 may not have fully penetrated the high-yielding fracture zone, and it may have lost efficiency since it was drilled.

Taking pumping test data alone into account when considering the aquifer's suitability to receive recharge water based on permeability, the borehole 9/16 area does not seem to be as suitable as the area around borehole 9/8A.

Because the borehole logs are not available, it is not possible to tell whether the aquifer is highly fractured above the water level. The opinion, prior to the injection test, was that if the unsaturated zone contains areas of high permeability, and if the design of the borehole did not significantly limit flow, then borehole 9/16 would prove to be a good injection borehole.

In terms of available storage, the suitability of the borehole would depend on the permeability and porosity of the unsaturated zone. The starting water level was only 9 m below the 1985 pre-pumping water level (Figure 26); thus, unlike the aquifer at borehole 9/8A, significant space had not been created by pumping. For the aquifer to accept artificial recharge, the injected water would need to occupy the unsaturated zone, or it would have to move down the hydraulic gradient towards the water level depression around borehole 9/8A.

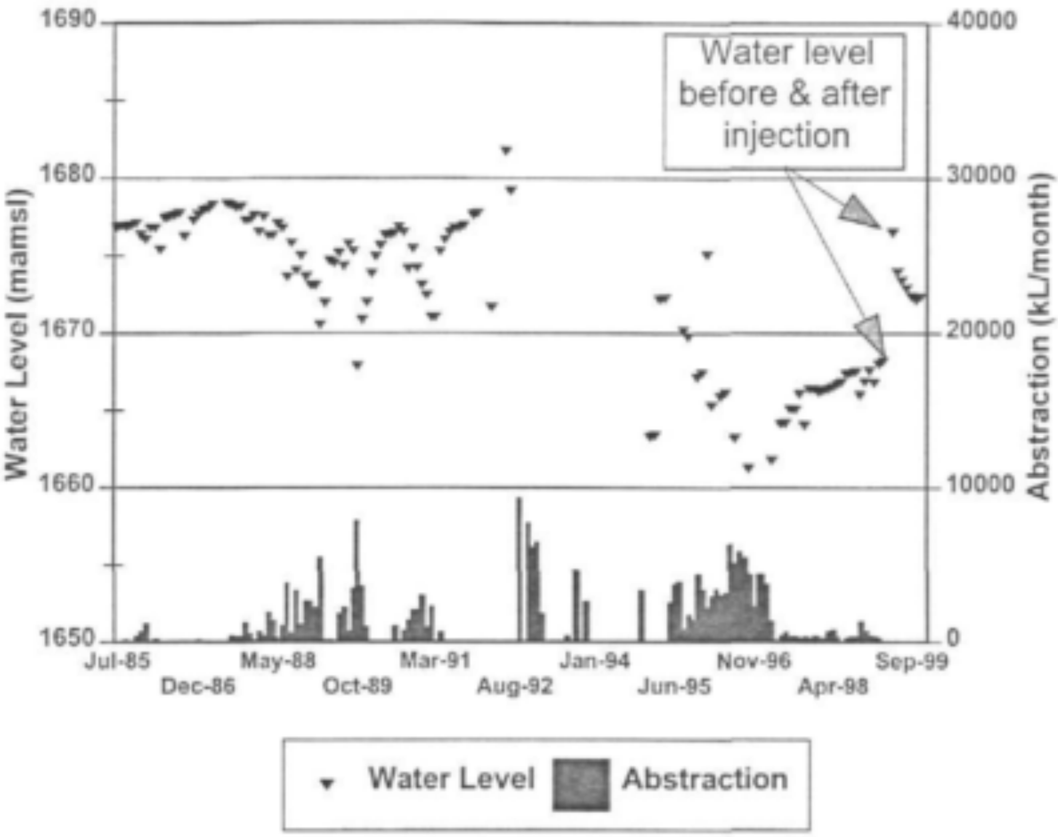


Figure 26 Water levels and abstraction data from borehole 9/16

6.2 The injection test

The maximum available gravity flow to this borehole was 5.9 L/s. Two steps were done at injection rates 3.6 L/s and 5.9 L/s (Figure 27). The last step was extended for four days to establish whether the water level would stabilise or continue to rise. The water level appeared to stabilise around 46 m above the original water level, indicating that at this depth, either a significant fracture zone exists, or that this was the required pressure head to get water to steadily flow into the fracture zone at 120 m. The recovery test (Figure 28) did not show signs of significant fracturing above the original water level. Because the water level stabilised at the end of the step injection test, it was decided that a higher injection rate should be used for the constant injection test. An in-line booster pump was installed in order to obtain a higher injection rate, with the hope that the borehole could receive a higher flow.

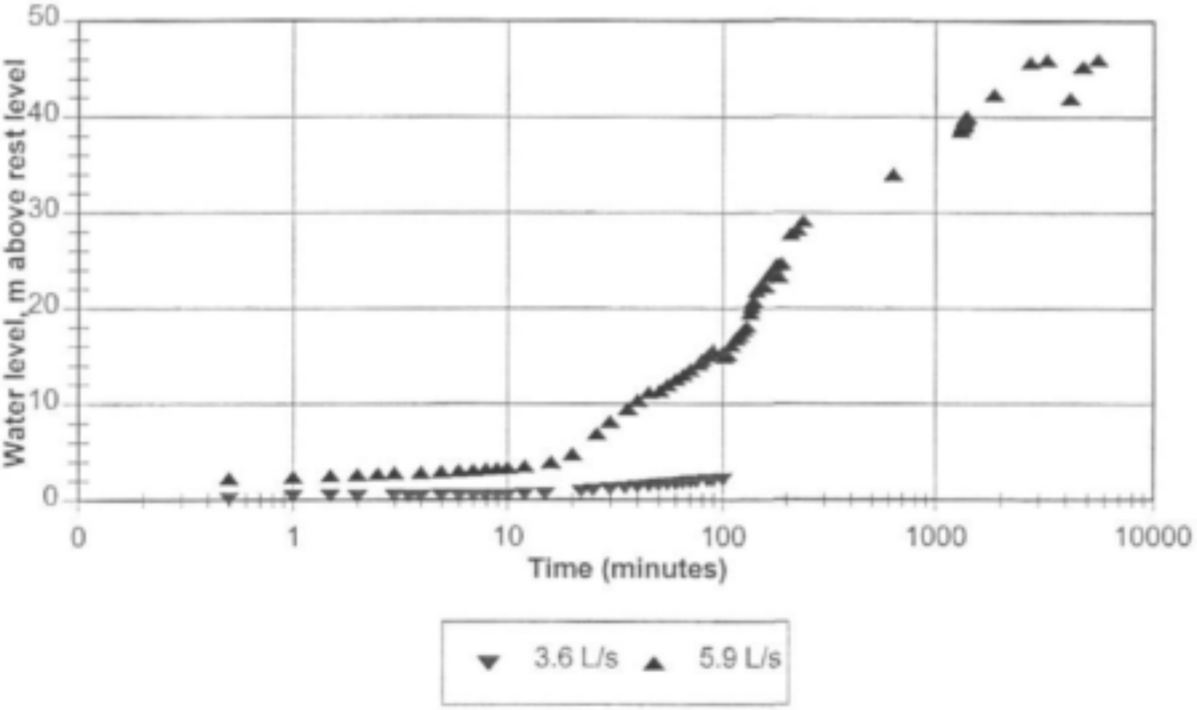


Figure 27 Borehole 9/16: Step Injection Test (Rest Water Level = 106.31 m below surface)

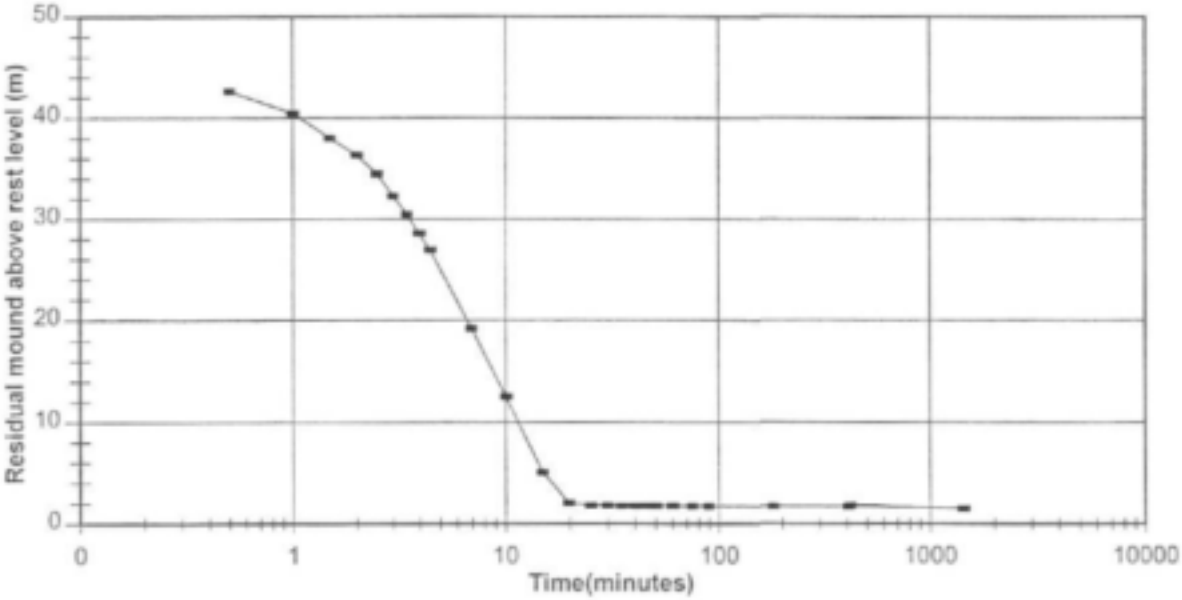


Figure 28 Artificial recharge mound dissipation following the step injection test

The **long term injection test** ran for 45-days (Figure 29). The volume of water injected, at an average rate of 6.2 L/s, was 24 025 m^3. The injection rate started at 13 L/s, but had to be reduced due to the rapid water level rise within the borehole.

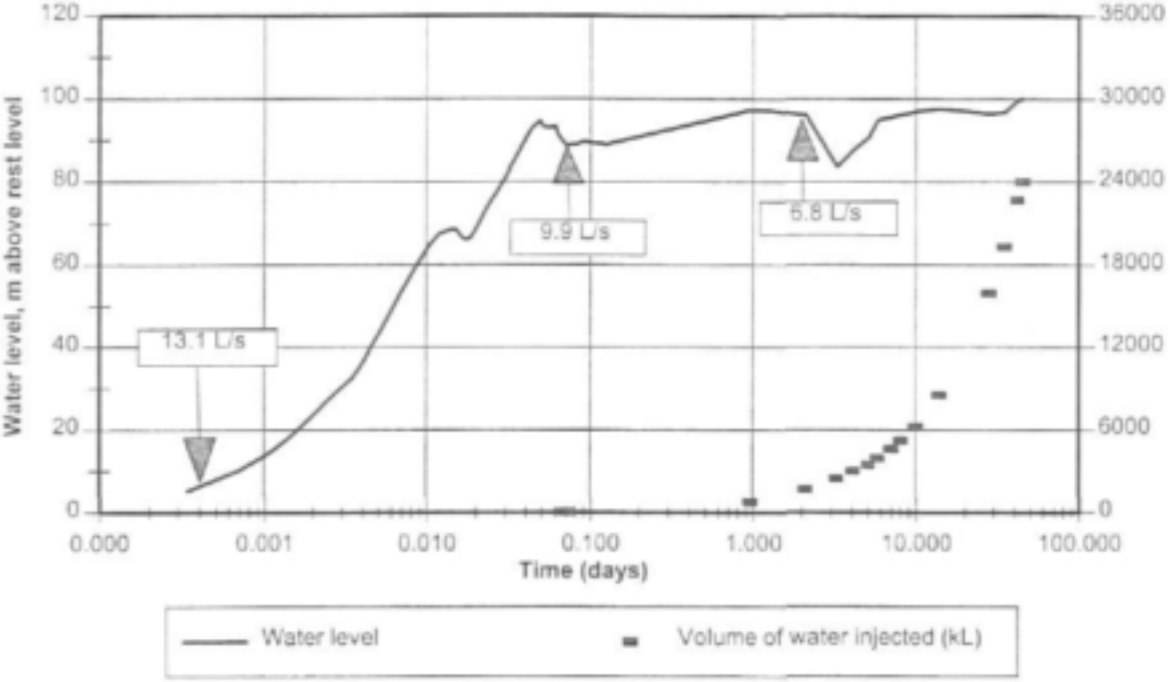


Figure 29 Borehole 9/16: Long Term Injection Test (Starting Water Level = 104.57 m below surface; Average injection rate = 6.2 L/s)

6.3 *The regional effect of injection*

Figure 30 shows that only borehole 9/17, which is 410 m from borehole 9/16, responded to injection. Borehole 9/17 is the only borehole which could not be taken out of production for this test. The two lowest readings (prior to injection) in Figure 30 reflect this borehole's pumping water levels. The average volume pumped from borehole 9/17 is 30 m^3 to 35 m^3 per week, in comparison to 3 730 m^3 per week injected into borehole 9/16.

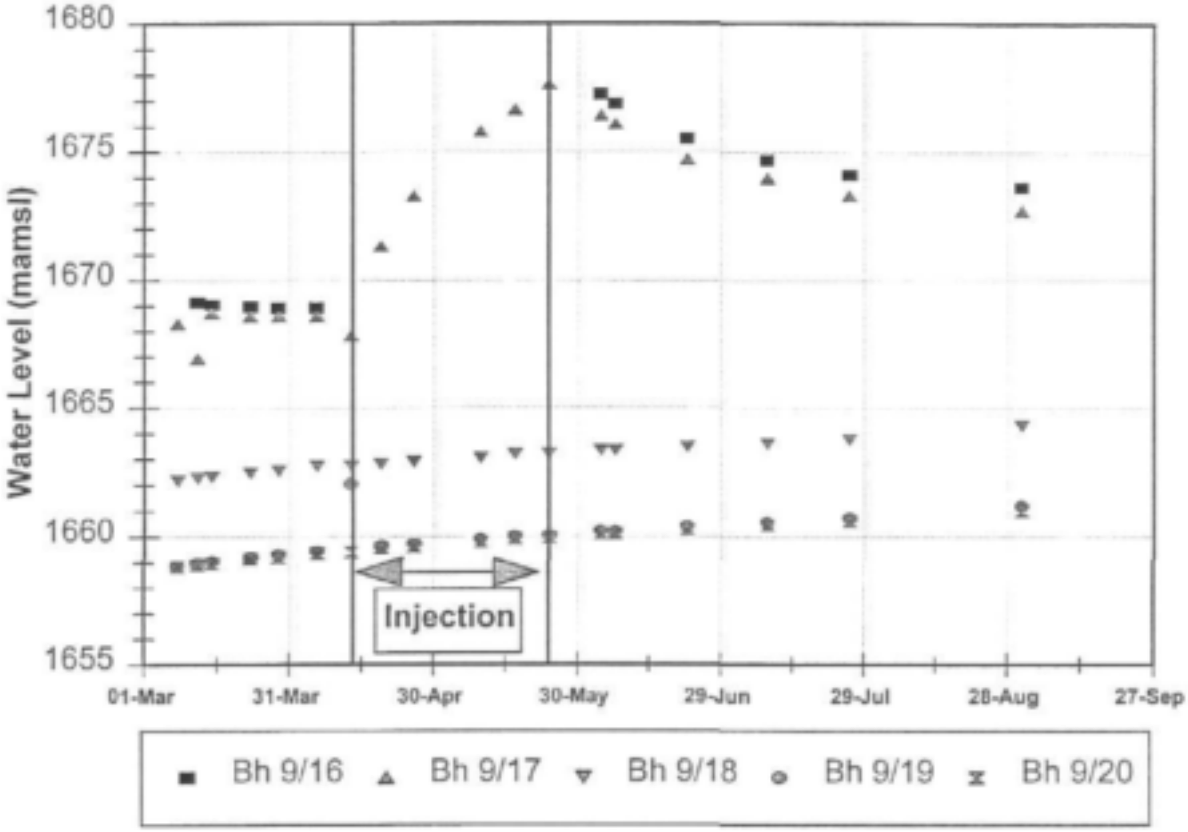


Figure 30 Water level response to injection in borehole 9/16

6.4 Aquifer permeability and storage

The transmissivity value obtained from borehole 9/17 using injection curve data is similar to the value obtained using pumping test data. Injection data gave a T-value of 35 m^2/day, and pumping data, 44 m^2/day. The S-values differed somewhat: the injection data gave a value of 0.001, and the pumping data, 0.0005.

These relatively low values could possibly be ascribed to the boreholes not fully penetrating the fractured quartzites in this area. Borehole 9/17A, drilled in close proximity to borehole 9/17, had to be abandoned because of the intense fracturing that was encountered during drilling. Using air percussion, it was not possible to fully penetrate the fracture zone.

Although the T- value obtained from borehole 9/17 is relatively low in comparison to the borehole 9/8A area, this part of the aquifer, with the sizeable Venusberg Fault running though it, may be far more permeable than the observed data suggests.

6.5 Water quality issues

The pipeline supplying the injection water had to be back-flushed to get rid of the scale within the pipeline. The injection water is of high quality and of lower TDS than the groundwater; it is not expected to cause any chemical precipitation or clogging (Figure 31).

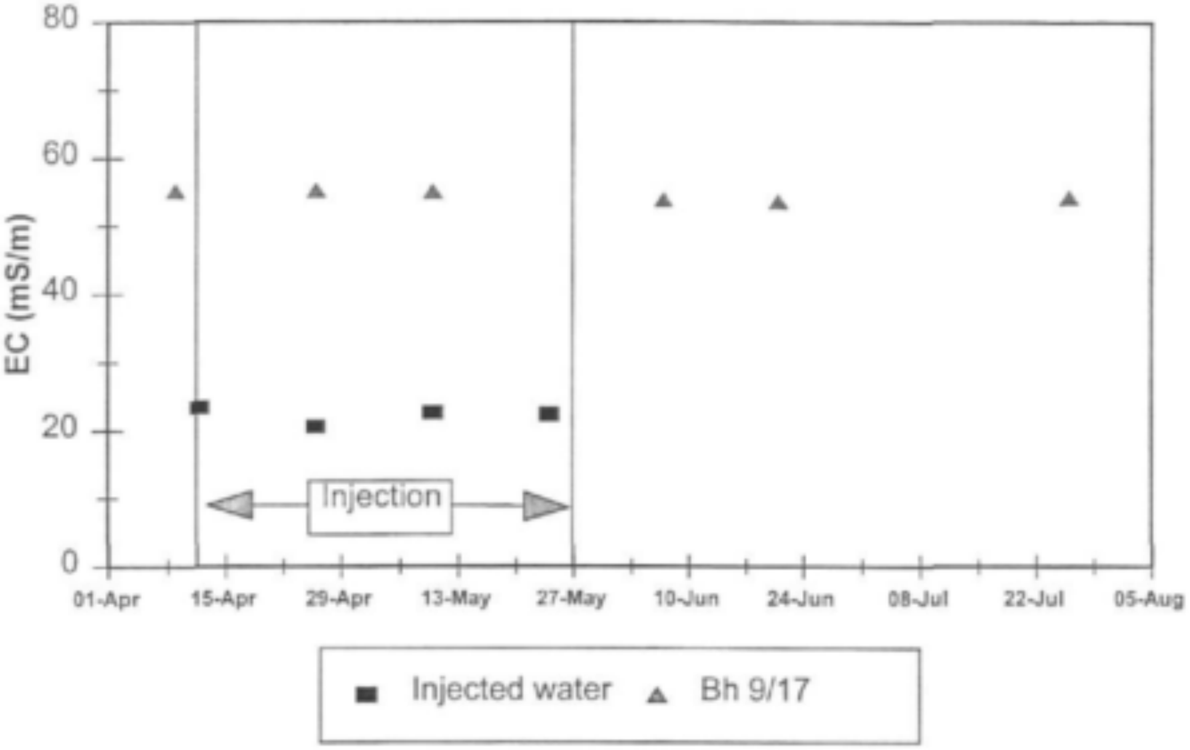


Figure 31 Electrical conductivity of the injected water and borehole 9/17 which is 402 m from borehole 9/16

6.6 Conclusions from borehole 9/16 injection test

This borehole can receive 500 m^3 per day, which is a third of what borehole 9/8A can receive. Although this is a good part of the aquifer to artificially recharge, a more suitably constructed borehole should be located or drilled in order to inject water at a much higher rate.

An alternative existing site would be borehole 9/20, which has a very similar drawdown curve to borehole 9/8A. Although borehole 9/20 was not constructed for injection, and is thus not an ideal injection borehole, the transmissivity value obtained for borehole 9/20 was 250 m^2/day, whereas at borehole 9/8A it was 60 m^2/day. A better option, however, would be to re-drill 9/17A with the aim of penetrating the fractured quartzites, since this site is further south than borehole 9/20 (and 9/16), and from past drilling it is known to be highly brecciated.

Figures 12 and 32 show that borehole 9/20 falls within a slight hydraulic depression between borehole 9/16 and borehole 9/18, and that the water levels drop off towards borehole 9/12. Water injected into the area between boreholes 9/20 and 9/17A should be easily recoverable from boreholes 9/16, 9/18, 9/19 and 9/20, since from the test pumping data, it is evident that there is good hydraulic connection between these boreholes; alternatively, the injected water could be recovered from the Group 9 boreholes located down-gradient to the east or north-east of the borehole 9/17 - 9/20 area.

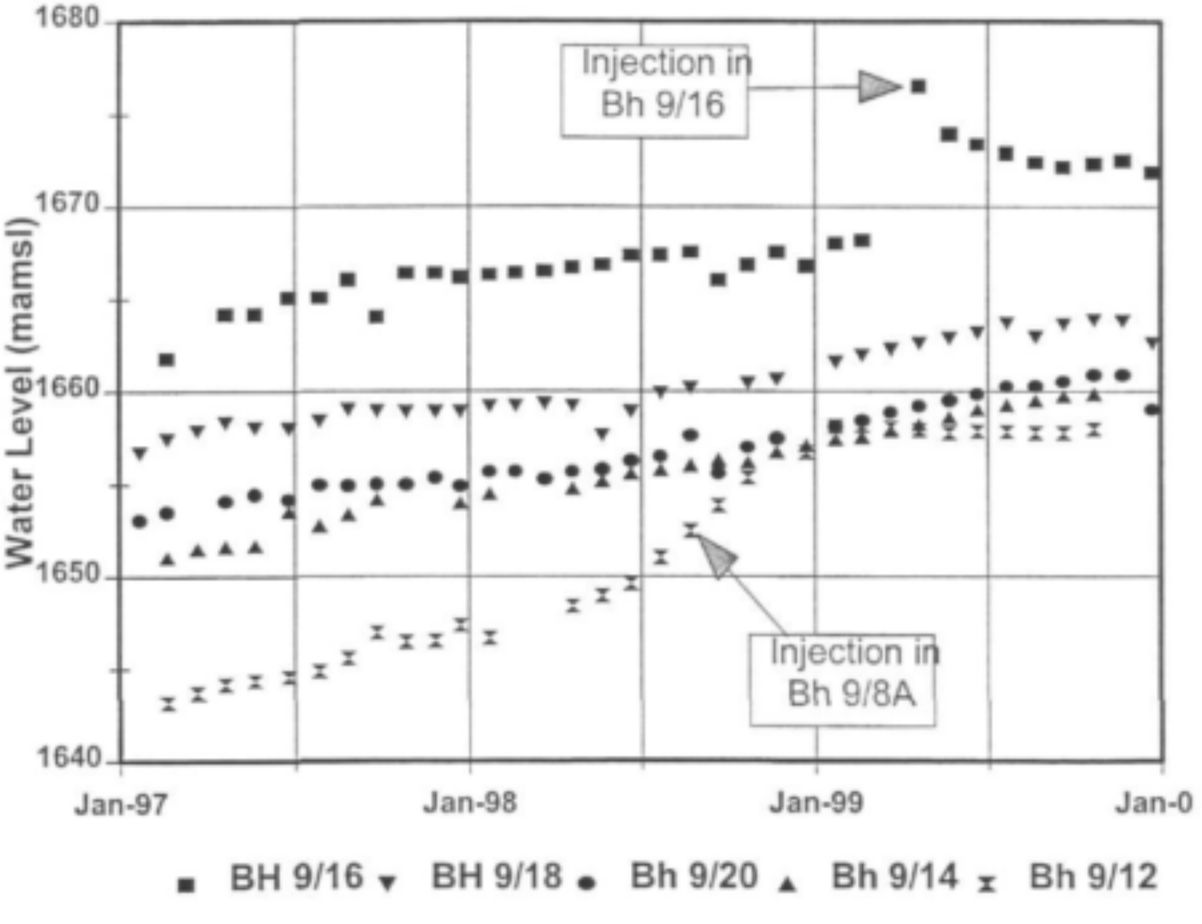


Figure 32 Water level relationship between boreholes north and north-east of borehole 9/16

7. BOREHOLE INJECTION TEST ON BOREHOLE 12/3

Borehole 12/3 is located in a highly transmissive part of the aquifer. Like borehole 9/16 it is located up gradient of the water level depression around borehole 9/8A (Figure 12 and Figure 33), and was therefore identified as a suitable area for injection. The water level in borehole 12/3 was 22 m above that of borehole 9/8A at the start of borehole 12/3's constant rate injection test. Plate 4 shows the layout of the injection pipelines at the borehole. Two pipelines from separate sources were joined at the borehole in order to achieve the maximum injection rate during the step injection test.

7.1 Borehole 12/3: Suitability for injection

This borehole was drilled in 1981 into quartzites and schists to a depth of 180 m. It is cased with 227 mm steel casing to a depth of 122 m. Water was struck at 108 m, 145 m and 165 m. The drilling yield of this borehole was 20 L/s and during the constant discharge test conducted in 1981 at 24.0 L/s, the drawdown was 5.1 m.

The borehole had to be unblocked and re-developed prior to the injection tests. The step injection test was attempted three times. During the first two attempts, the injection rates were too low. A booster pump was installed in addition to the gravity supply from the reservoir, and this allowed for a total of 59.4 L/s to be injected during the final step of the third attempt.

In terms of aquifer suitability to receive recharge water based on permeability, this part of the aquifer is far more suitable than the area around borehole 9/8A. After the borehole was unblocked, it was pumped for six hours at 18.6 L/s. From the recovery curve, a transmissivity value of 3 800 m^2/day was obtained using the Cooper-Jacob method. While this value may seem exceedingly high, it may well be reasonable in this highly fractured aquifer.

This area should also be suitable for artificial recharge in terms of available storage. Decades of pumping from boreholes north of this area has induced groundwater flow from the Auas Mountains towards the City of Windhoek, thereby creating space in the Group 12 area (Figure 34). Even though water injected into this area would occupy local storage, it will also flow down gradient and eventually fill the space that has been created in the hydraulic depression around borehole 9/8A.

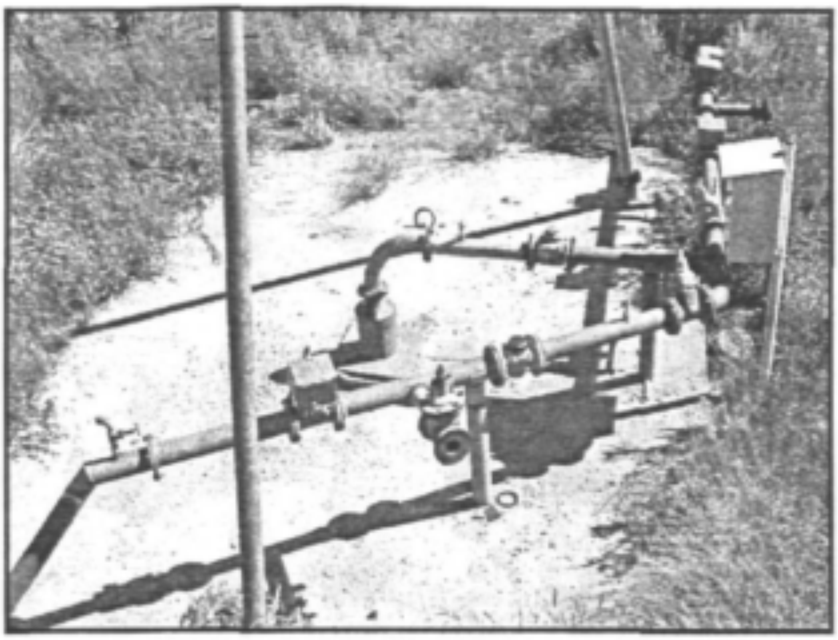


Plate 4 Borehole 12/3. Due to the high injection rate at this borehole, two pipelines had to be joined (top right and bottom left of the photo). Water flowed under gravity from the Luipaards Vallei Reservoir, and was pumped from the Kleine Kuppe Reservoir.

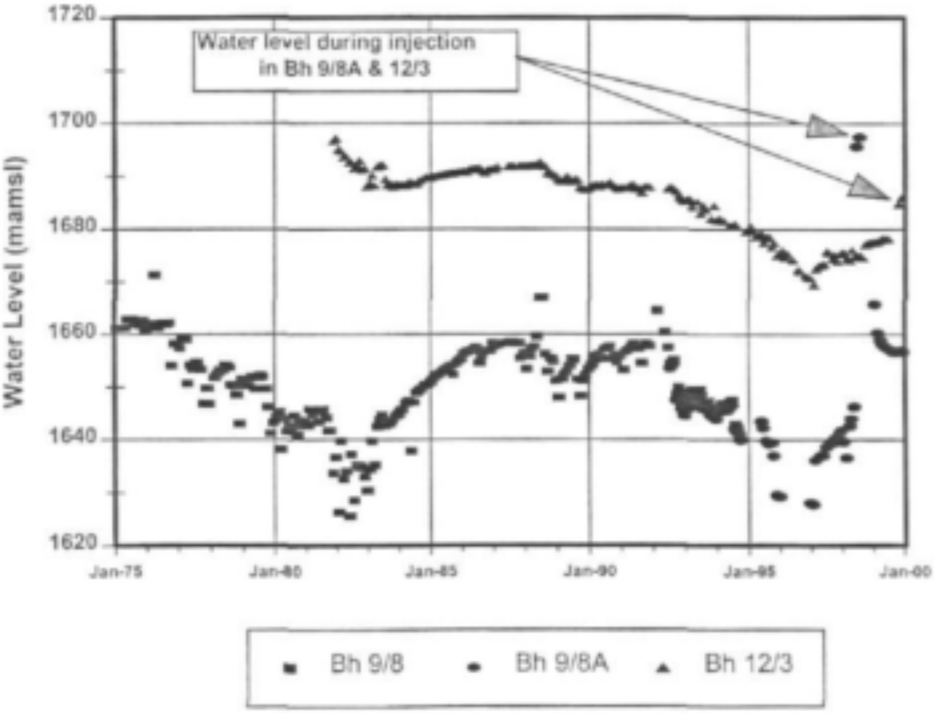


Figure 33 Water Levels in boreholes 9/8 and 12/3 showing the hydraulic gradient towards borehole 9/8

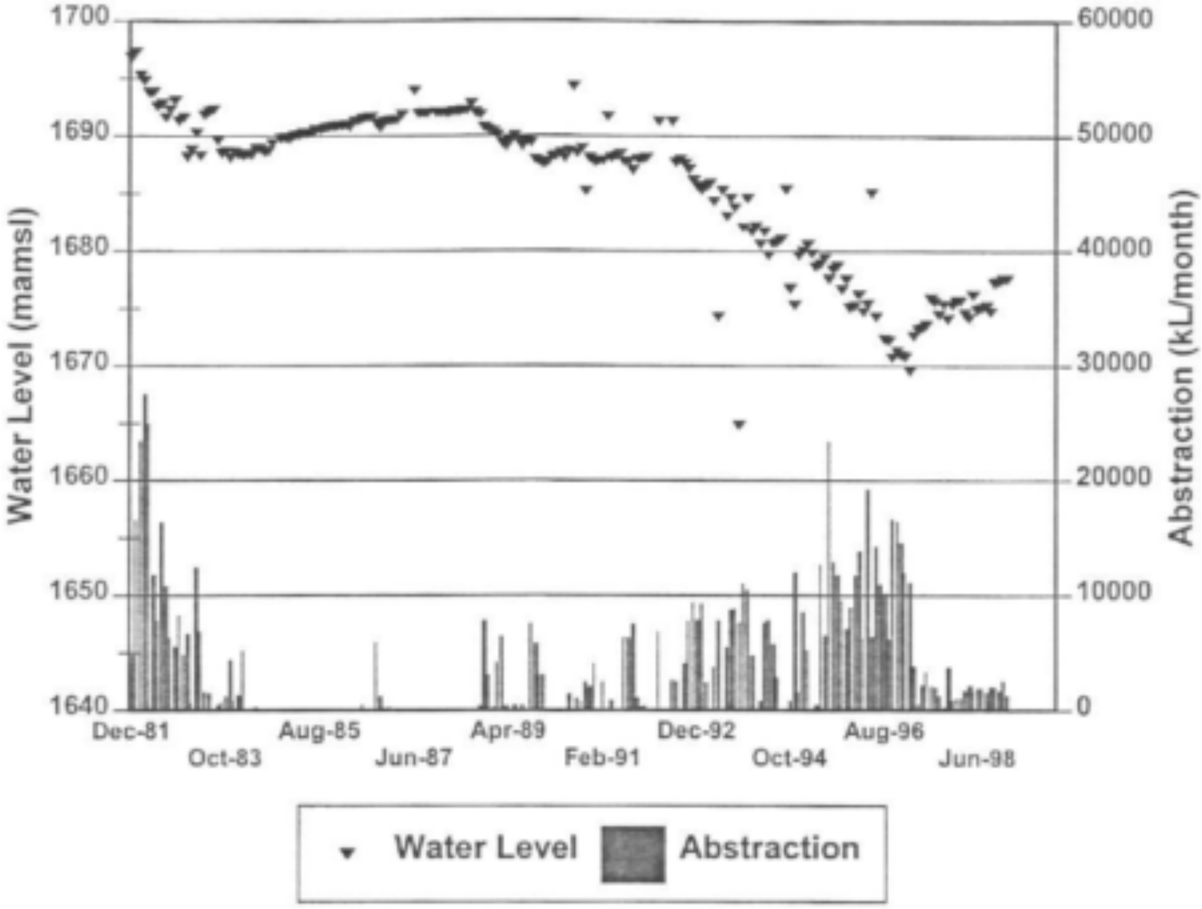


Figure 34 Water level and abstraction data from borehole 12/3

7.2 The injection test

Figures 35 and 36 show the results of the **step injection test**. Figure 35 confirms the aquifer's high permeability in this area. The last step was done at an injection rate of 59.4 L/s and the water level did not continue to rise as is normally the case.

It was not possible to do the 35-day **constant injection test** at 59.4 L/s since this would have required booster pumping at a high rate for the duration of this test. Rather, gravity flow from the supply reservoir was used for the test. This gave an average rate of 32.7 L/s. Figure 37 shows the water level rise in this borehole during the constant injection test.

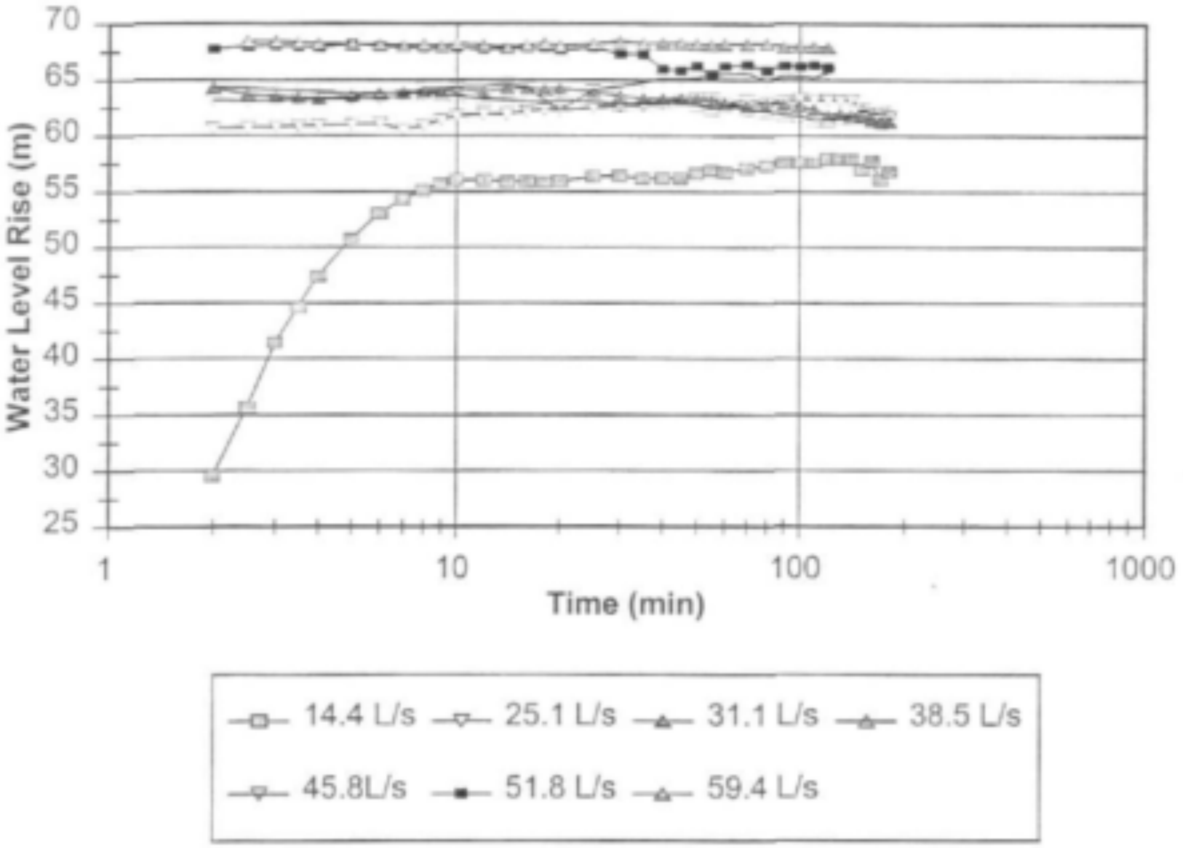


Figure 35 Borehole 12/3: Step Injection Test (Rest Water Level = 102.79 m below surface)

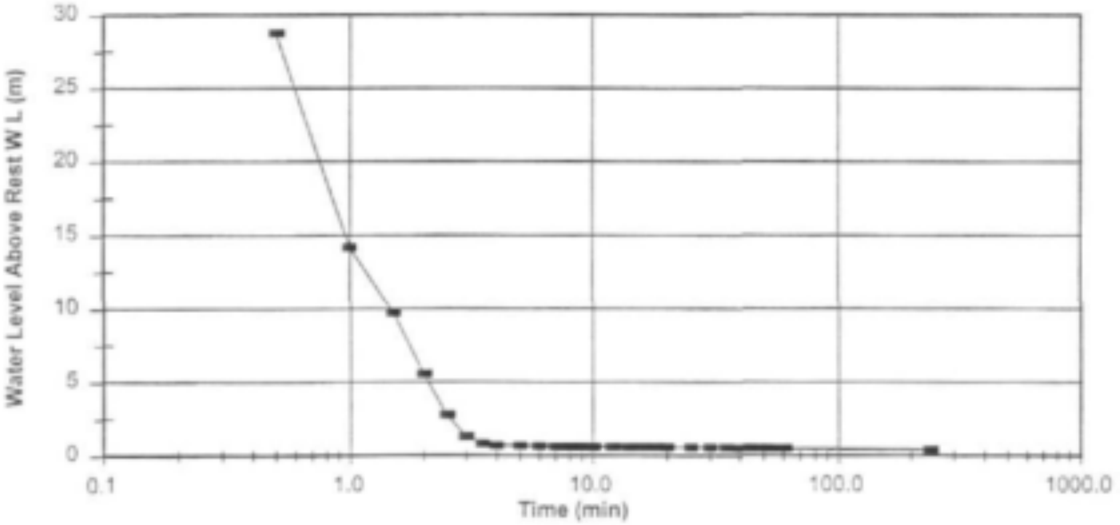


Figure 36 Artificial recharge mound dissipation following the step injection test

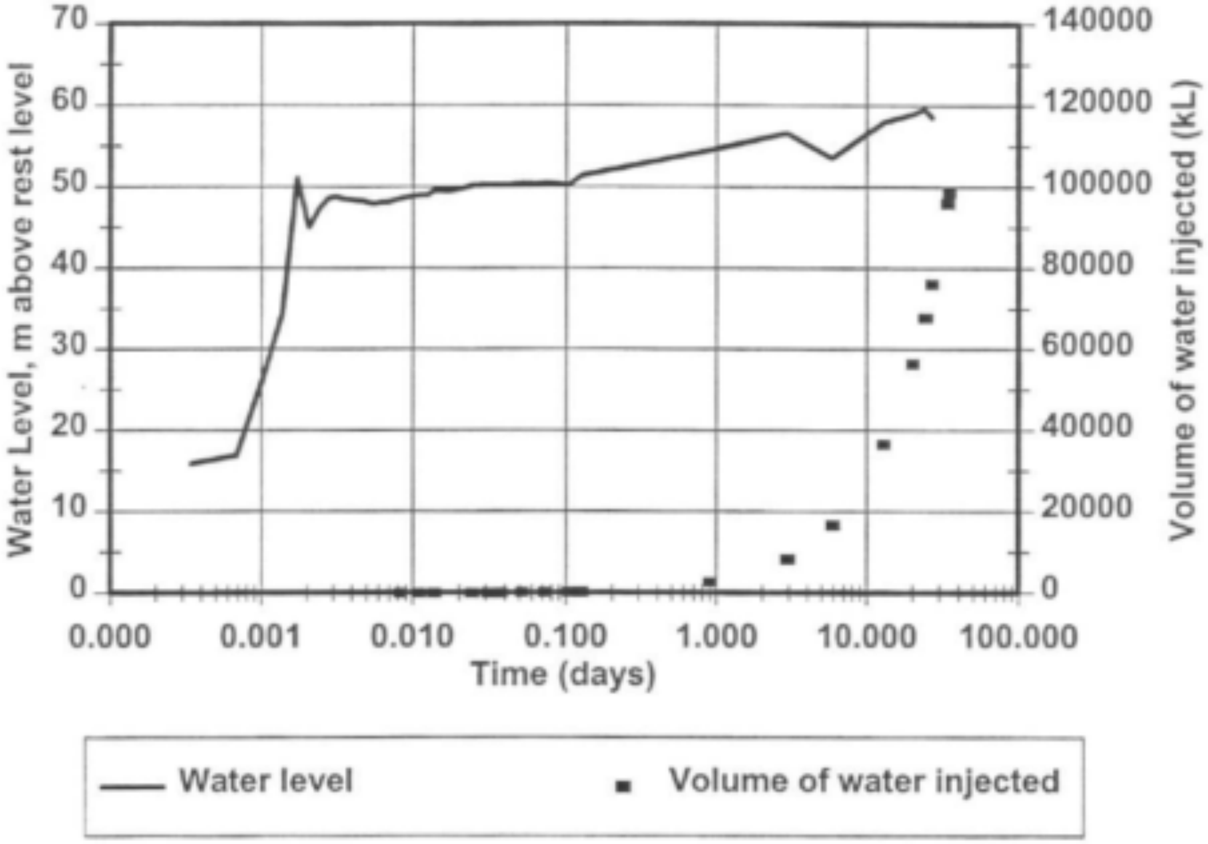


Figure 37 Borehole 12/3: Constant Injection Test (Starting Water Level = 102.8 m below surface; Average injection rate = 32.7 L/s)

7.3 The regional effect of injection

Figure 38 shows the water level response in boreholes around borehole 12/3. As in the previous injection tests, the response was remarkably quick. Boreholes 12/1A, 12/2 and 12/4 which are 1 348 m, 459 m and 1 133 m respectively from the injection borehole, responded within hours of injection (Figures 39 and 40). Like all other boreholes within the wellfield which have been monitored with pressure transducers, these boreholes also show the sinusoidal water level fluctuations which, together with the rapid water level response, indicate that the aquifer behaves in a confined manner.

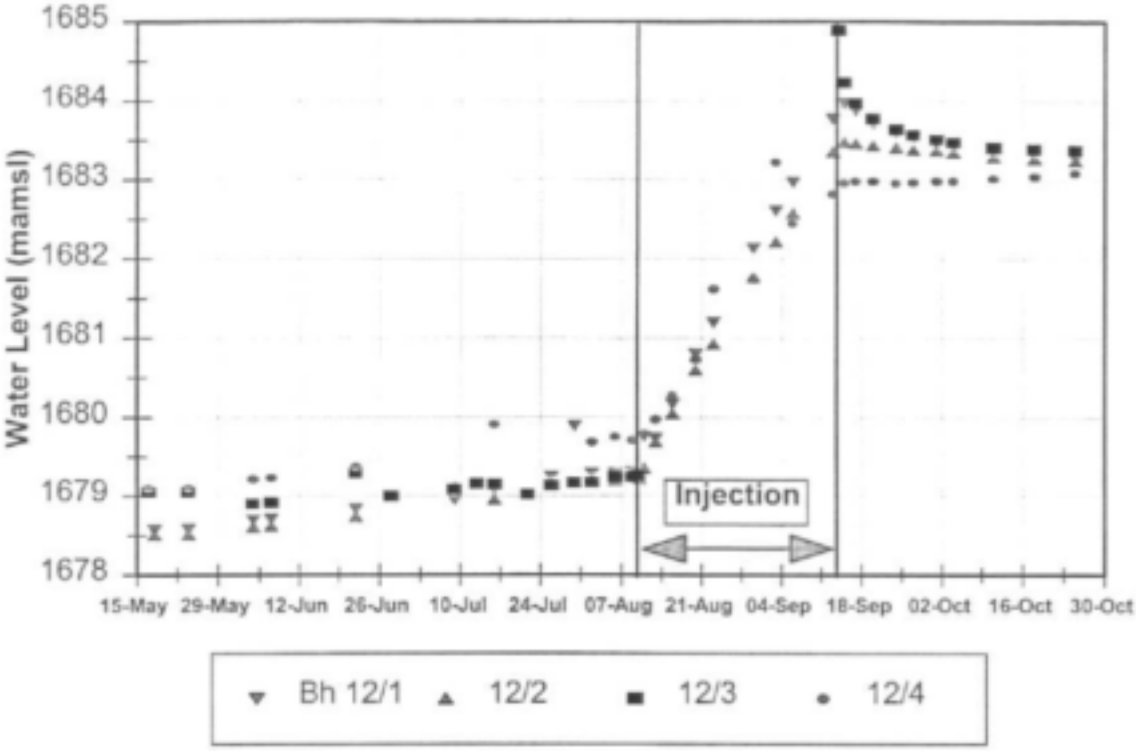


Figure 38 Water level response to injection in boreholes east and south-west of borehole 12/3

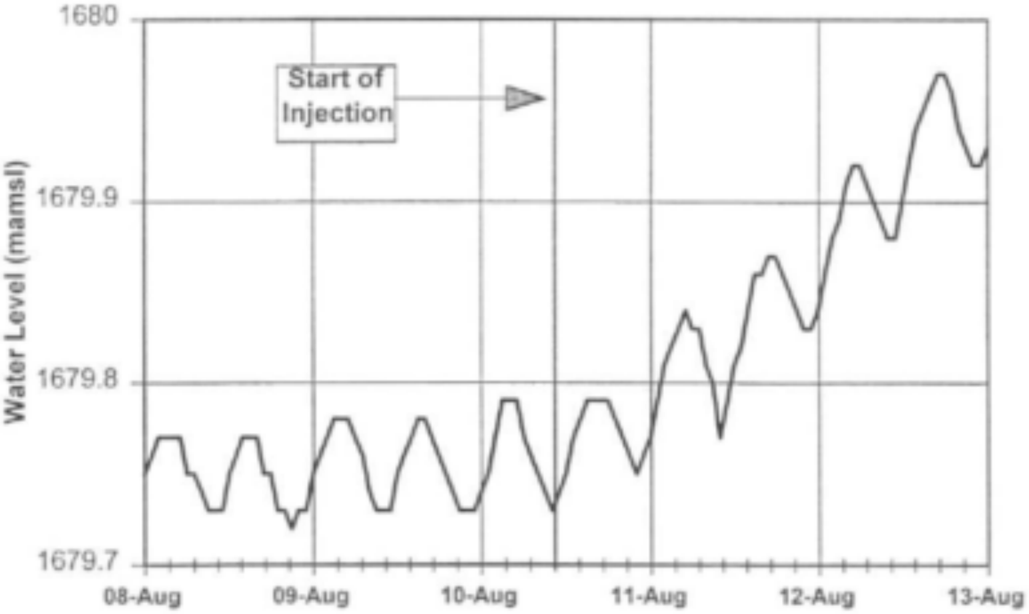


Figure 39 Borehole 12/4: Water level response to injection in borehole 12/3 showing the rapid response to injection and the effect of earth tides (Distance between borehole 12/3 and 12/4 = 1132.7 m)

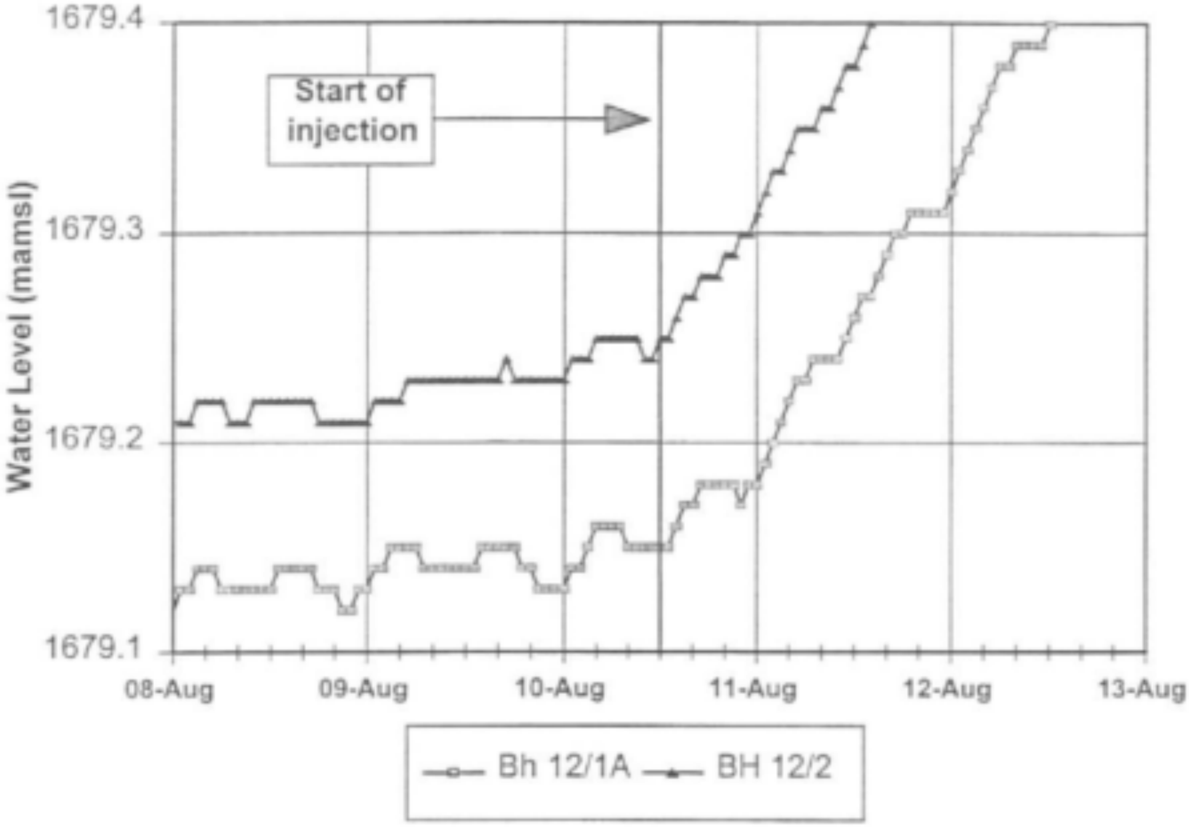


Figure 40 Borehole 12/1A and 12/2: Water level response to injection in borehole 12/3 showing the rapid response to injection (Distance between borehole12/3 and 12/1A = 1347.8 m and 12/3 and 12/2 = 458.8 m)

After the injection test on borehole 12/3, it was discovered that the water level in borehole 9/13 could again be measured. This was not the case in May 1998 when all Group 9 borehole water levels were measured prior to injection in borehole 9/8A. Borehole 9/13 has a blockage at about 125 m, but some time after mid-1998 the water level rose above the blockage, and readings could once again be taken. For this reason, it was decided to run the injection test on borehole 12/3 again in order to monitor the response in borehole 9/13. This test was conducted for ten days at 38.9 L/s and borehole 9/13 responded to injection after approximately six days (Figure 41).

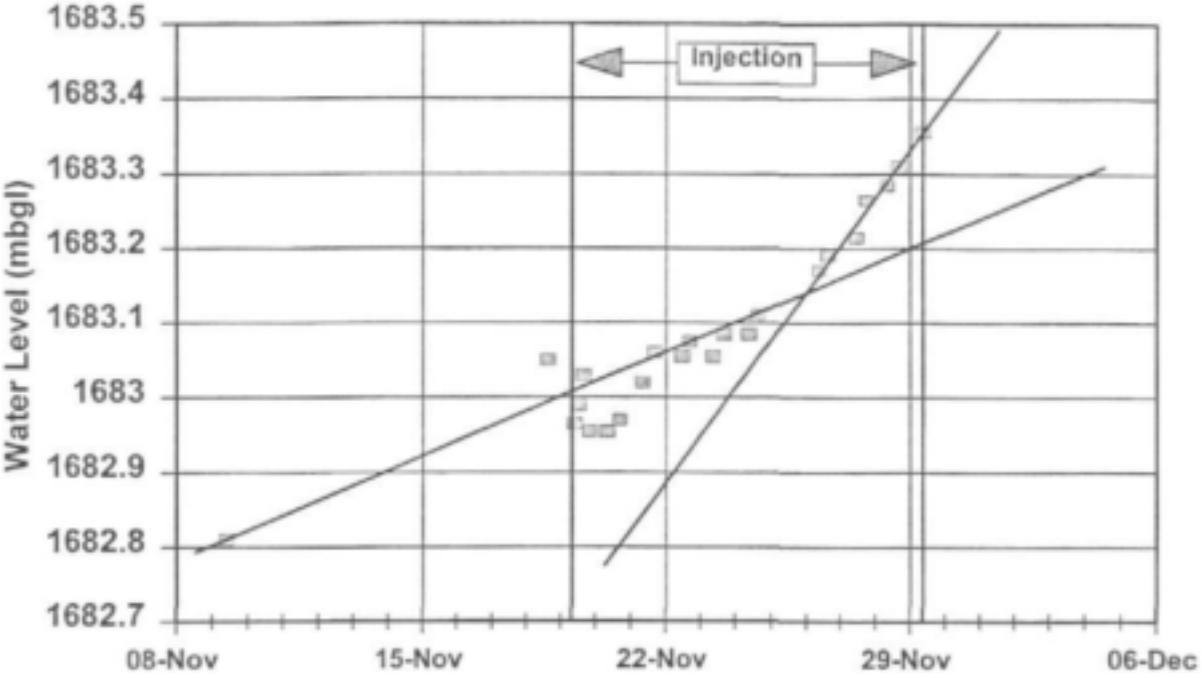


Figure 41 Water level response to injection in borehole 9/13, 591 m north-west of borehole 12/3

7.4 *Aquifer permeability and storage*

Pumping test data from boreholes 12/3; 12/1 and 12/1A gives the following average aquifer parameter values:

T-fracture: 1 600 m^2/day (12/1 and 12/1A) to 4 600 m^2/day (12/3)
T-formation: 540 (12/1 and 12/1A) to 1 500 m^2/day (12/3)
S-formation: ± 0.005

The injection curves from boreholes 12/1A; 12/2 and 12/4 gave the following values:

T-fracture: 1 600 m^2/day
T-formation: 80 m^2/day (this value appears to have been affected by boundary conditions)
S-formation: ± 0.003

The S-value was reassessed after considering the volume of water injected in relation to the volume of rock mass saturated by injection (Figure 42).

The calculated S-value is:
= 0.099 Mm^3 / 11.4 Mm^3
= 0.0087

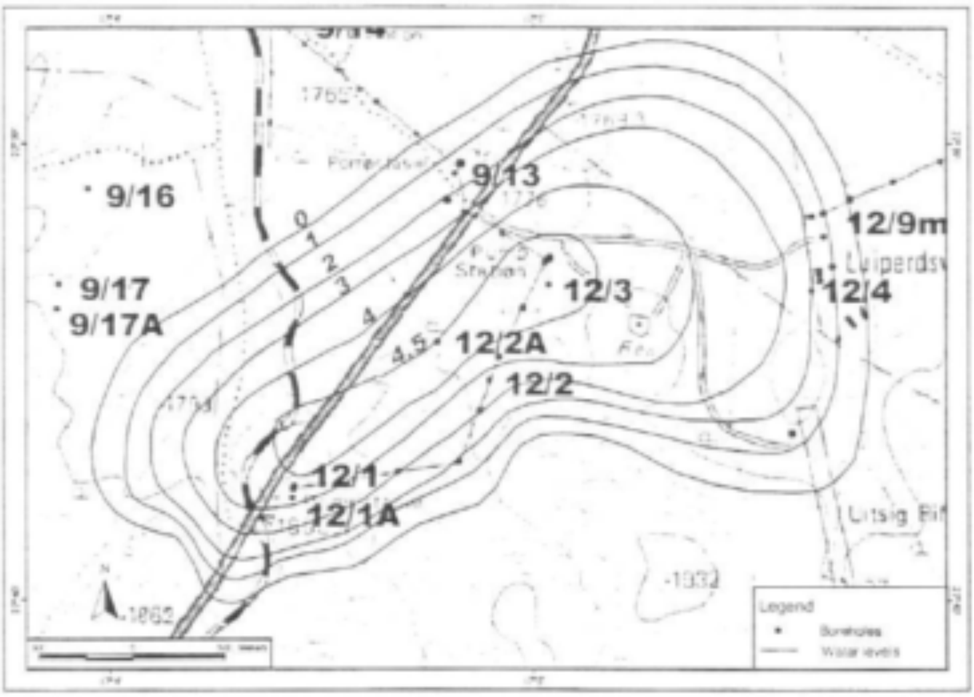


Figure 42 Water level rise as a result of injection into borehole 12/3

Both the pumping and injection test data show that the aquifer in this area has an extensive network of highly permeable fractures. In these highly fractured quartzites, the S-value of 0.009 obtained from the water level rise contours could well be more realistic than the 0.003 to 0.005 obtained from the pumping and injection test curves.

7.5 Water quality issues

Water quality was monitored at three sites: Electrical conductivity was monitored at borehole 12/1A with a data logger; electrical conductivity and pH were monitored at borehole 12/2 with a data logger; and water samples were collected from borehole 12/1 and analysed by the Municipal Laboratory. Figure 43, based on samples analysed in the Municipal laboratory, shows that the changes in electrical conductivity and in the sulphates at borehole 12/1 were minimal. The electrical conductivity and chlorides (Figure 44) at borehole 12/1 appear to follow a similar trend compared to the injection water. This trend does not necessarily mean that injection water has moved to borehole 12/1 since the concentration changes are small, and close to the bottom end of the scale where analytical precision may be insufficient.

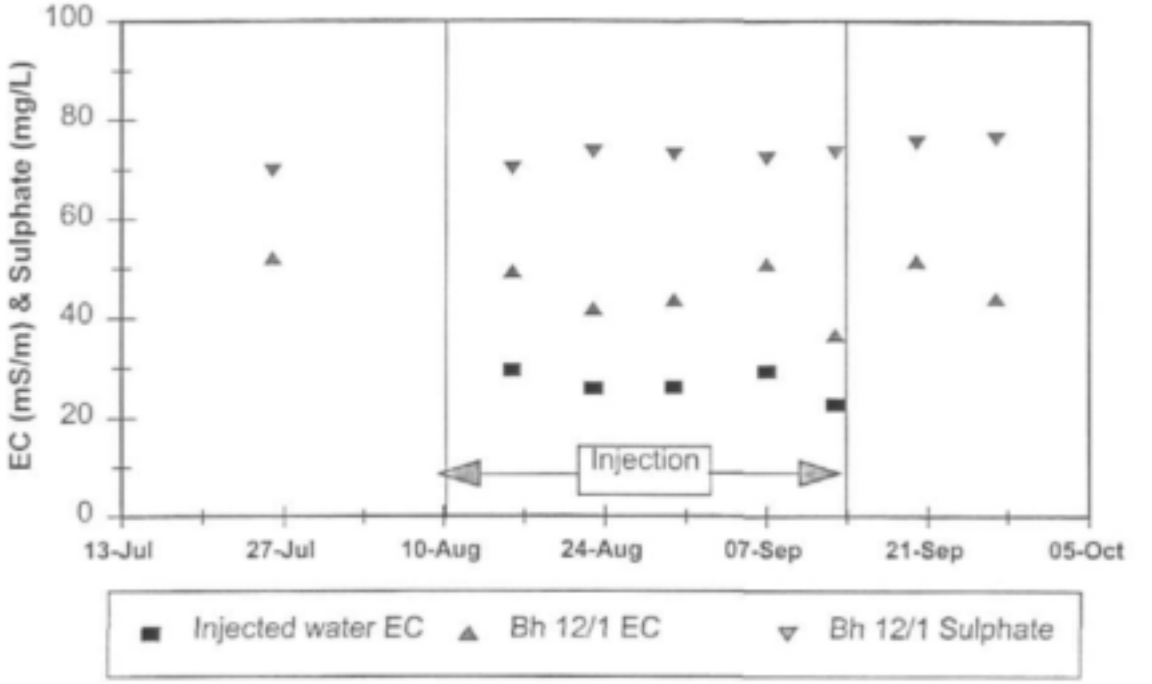


Figure 43 Electrical conductivity of the injected water at borehole 12/3 and electrical conductivity and sulphate at observation borehole 12/1

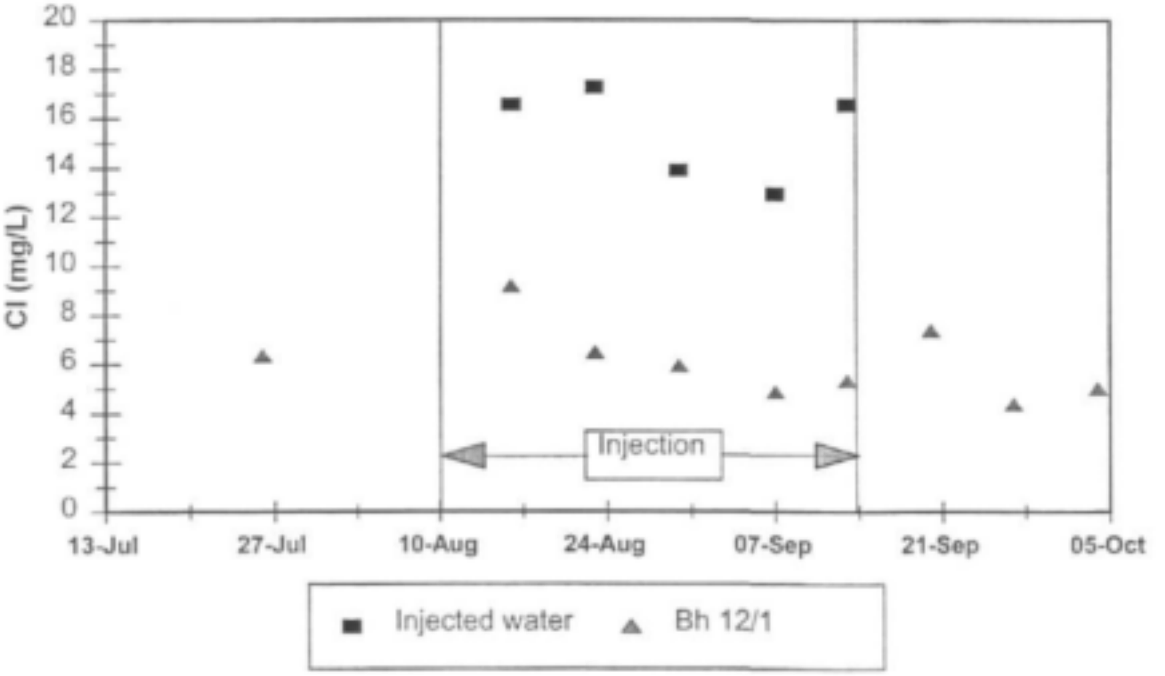


Figure 44 Chloride of the injected water at borehole 12/3 compared with chloride at observation borehole 12/1

The data loggers in boreholes 12/1A and 12/2 present interesting time series' of data. In both cases the sensors were installed a metre below the pre-injection water levels. Figures 45 and 46 show how the salinity and pH changed during and after injection. Figure 36 shows that once the water level in the borehole rose above 1681.9 m, it's salinity started to increase. This response is similar to that of borehole 9/11 after injection into borehole 9/8A.

The explanation offered is that sulphate minerals were present in the upper, dewatered portion of the aquifer. As the water level rose in the vicinity of borehole 12/1A, these salts were dissolved and this caused the increase in salinity at borehole 12/1A. Gradually these salts were leached as the water slowly migrated past the borehole. For this reason a peak appeared which gradually dissipated once the injection test was stopped. In this case it is assumed that had the test continued for a longer period, the salinity would eventually also have decreased.

In the case of borehole 12/2, the electrical conductivity response occurred once the water level had risen to it's peak of just above 1683 m at the time injection stopped. This response is difficult to explain; it could relate to the physical design of the borehole, or it may be result of slow downwards flow after the water had time to pick up salts in the uppermost part of the aquifer in the vicinity of borehole 12/2.

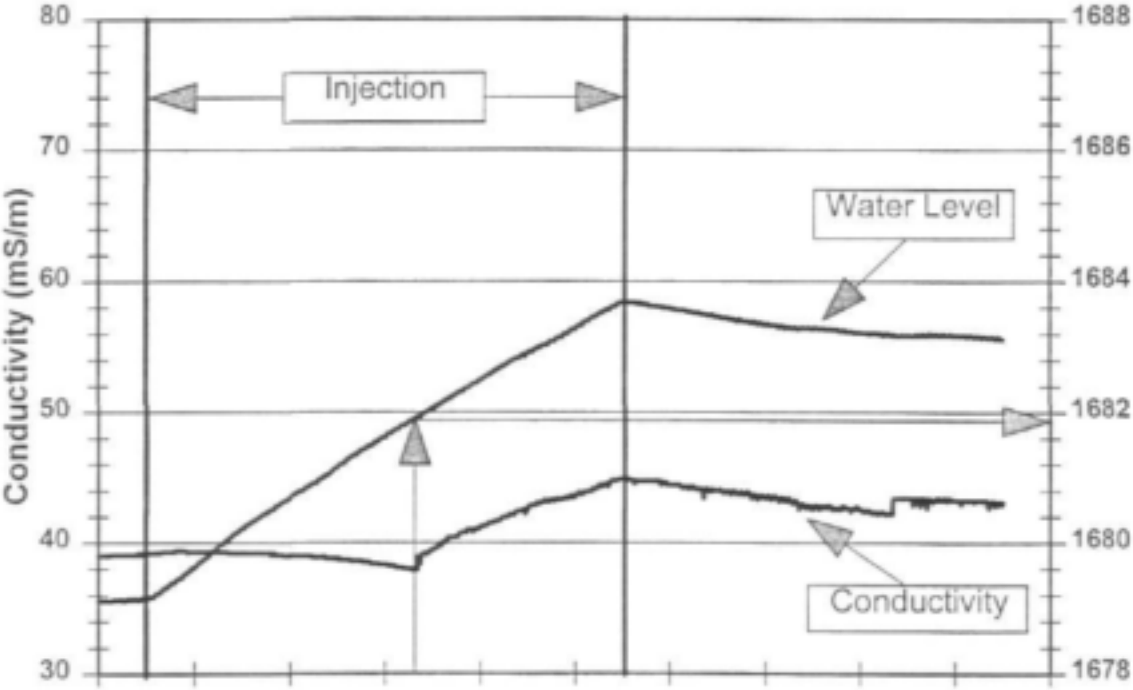


Figure 45 Electrical conductivity and water level response in borehole 12/1A as a result of injection

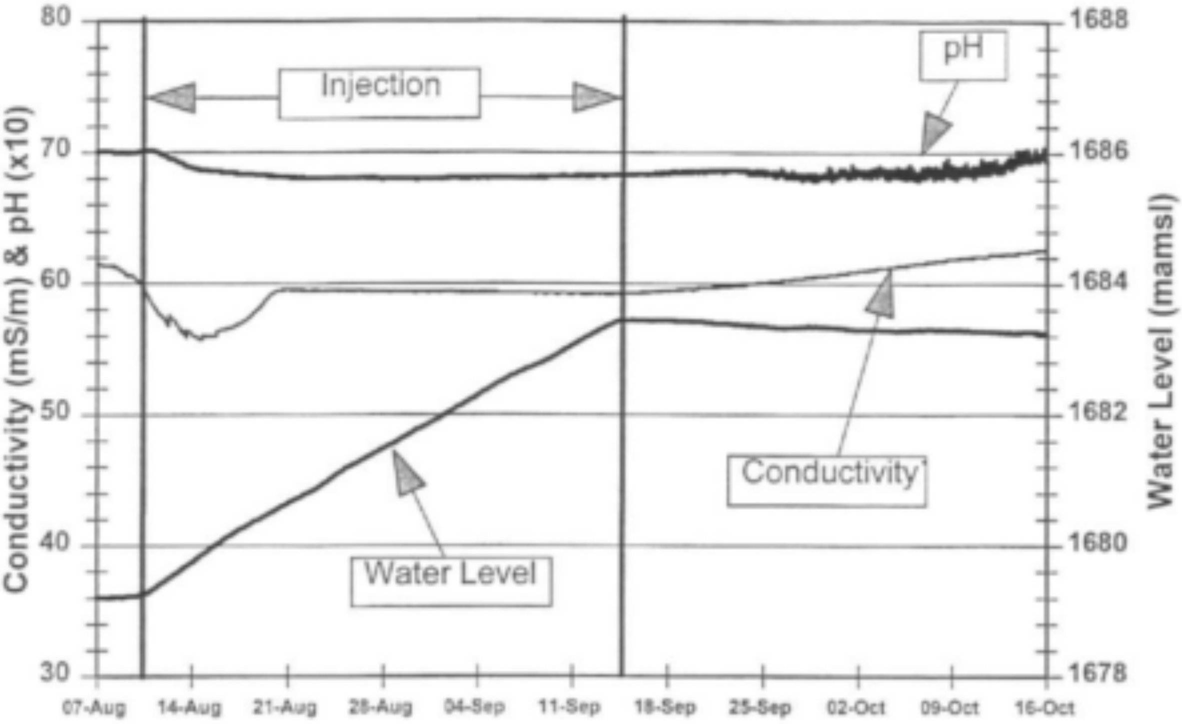


Figure 46 Electrical conductivity, pH and water level response in borehole 12/2 as a result of injection

7.6 *Conclusions from borehole 12/3 injection test*

This borehole was tested at a maximum injection rate of 59.4 L/s (5 132 m^3/day) during the step injection test. The constant injection test was run at 32.7 L/s or 2 825 m^3/day, which is double the injection rate obtained at borehole 9/8A. Even though this borehole was not designed for injection, it is suitable for use as an injection hole. Under gravity flow conditions with existing water supply infrastructure, a long-term injection rate of 1 Mm^3/a (2 740 m^3/day) should be achievable.

The aquifer in this area is highly transmissive and most suitable for artificial recharge. In relation to other boreholes within the Windhoek aquifer, the Group 12 boreholes on the northern slopes of the Auas Mountains are the most suitably located boreholes for long-term injection. This is because they penetrate the highly permeable and more pure (less micaceous) Auas Formation quartzites, and because they are situated closest to the main natural recharge and storage areas - the northerly dipping quartzites of the Auas Mountains. This part of the Windhoek aquifer can receive water at rates far in excess of the 1 Mm^3/a that the current water supply infrastructure can provide. Existing boreholes, in addition to borehole 12/3 that should be investigated for use as potential injection boreholes are 12/1A, 12/4, 12/9, and the eastern most boreholes, 12/5 and 12/6.

8. CONCLUSIONS

This report concludes the pilot testing of artificial recharge by borehole injection into the Windhoek aquifer. Four borehole injection tests have been done to date. In 1997 an injection test was done on borehole 9/5 east of the Eros Airport under the supervision of Carr Barbour & Associates; and between July 1998 and September 1999 three borehole injection tests were done on the Group 9 and 12 boreholes south of the Eros Airport under the supervision of the CSIR.

In these tests, treated surface water was injected via deep boreholes into permeable parts of the aquifer. The longest test lasted 195 days and the volume of water that was injected was 289 000 m^3. The high injection rates of up to 59.4 L/s achieved during the tests are very encouraging and confirm the feasibility of a full scale injection scheme.

There has been a measurable effect on the Windhoek aquifer to injection. This was observed by the water level rise in observation boreholes up to 1.3 km from the injection boreholes. The next step in this project is to proceed the first phase of constructing a full scale artificial recharge scheme.

Out of the pilot injection boreholes, those which are most favourably located for artificial recharge are the ones south of the Eros Airport. These boreholes are closer to the main area of natural recharge in Auas Mountains and are further from the area of natural discharge from the Windhoek aquifer, which is north-west of Windhoek. Borehole 9/5 is less suitable for long term injection because it is located closer to the discharge area of the aquifer. This borehole is thus more suited as an observation borehole or as production borehole in times of need.

Assuming that borehole clogging as a result of injection is either not a problem, or that it can be adequately managed, the rate at which water can be injected into the three recently tested boreholes is 1.65 Mm^3/a (Table 6).

Table 6 continues...

Table 6 Injection volumes for boreholes 9/8A; 9/16 & 12/3

Borehole Number	*Injection Rate (M^3/hr)*	*Monthly Injection Volume (m^3)*	*Yearly Injection Volume (m^3/a)*
9/8A	60	43 000	520 000
9/16	20	14 000	170 000
12/3*	110	79 000	960 000
Total injection	190	136 000	1 650 000

12/3* The injection rate given is the rate at which water can flow under gravity from the supply reservoir. The step injection test indicates that the borehole can receive 213 m^3/hr, which is equivalent to 1.86 Mm^3/a.

The quality of the injection water is good and should not have any negative impact on the Windhoek aquifer. It is essential that poor quality water be prevented from entering the aquifer. In order to do so all future injection water should be fed through a granular activated carbon filter and be chlorinated prior to injection. It is also important to remove chemical deposits and silt in the reservoirs and injection pipelines prior to injection. The pipelines will need to be flushed until the water is absolutely clear.

9. RECOMMENDED IMPLEMENTATION PHASES

Three artificial recharge phases are recommended:

Phase 1:	Minor infrastructural modifications;
Phase 2:	Up-grade to a full-scale borehole injection scheme;
Phase 3:	Assess other areas within the Windhoek aquifer that may be suitable for artificial recharge.

By implementing Phase 1, the artificial recharge scheme will primarily address the problem of excessive drawdowns in the Group 9 wellfield after years of high abstraction from this area. The water level recovery in this area should be reduced from 5-6 years to less than 2 years if there is continuous injection.

By implementing Phase 2, the artificial recharge scheme will begin to address the problem of groundwater mining that appears to have taken place over the past 50 years. Only by implementing this phase, can the long-term reliability of the aquifer be guaranteed.

The implementation of Phase 3 will depend on the results of Phases 1 and 2. Should Phases 1 and 2 prove to be successful in relation to all the factors that determine the success of an artificial recharge scheme (including non-hydrogeologic factors such as economics and management), then other sites within the Windhoek aquifer should be considered for artificial recharge. An overview of the factors to consider is provided in Murray and Tredoux (1998).

Table 7 provides a summary of estimated artificial rates after completion of Phases 1 and 2 of the artificial recharge scheme.

Table 7 Estimated artificial rates after completion of Phases 1 and 2 of the artificial recharge scheme

Implementation Phase	*Possible Completion Year*	*Maximum Artificial Recharge Rate (Mm^3/a)*
Phase 1	2002	3
Phase 2	2004	5.5
Phase 3	?	10?

The following recommendations were made by Dr P Dillon after reviewing this project:

Water quality issues yet to be evaluated are: As, Radon and possibly DBP's (notably THM's). It is recommended that some preliminary monitoring of these parameters be included in project development to rule these out or to determine whether remedial actions, such as further treatment of injectant or recovered water are required. It is recommended that DOC (and TOC) be kept to as low a level as possible in injectant, in order to minimise microbial growth in the aquifer and thereby reduce production of biomass and extra-cellular polysaccharides (slimes) that could ultimately clog the flow pathways especially where these have small apertures. It also reduces the substrate available to support microbially mediated reactions, including those redox-related reactions which could influence water quality, such as sulphate reduction.

Given the cost of GAC treatment and rechlorination, a long-term trial undertaken with and without this treatment train may be warranted. Such a trial would demand the presence of at least one observation well in close proximity to the injection well, in order to assess any increase in well losses which may be attributed to microbial action.

Dr Dillon also recommended developing a numerical model to assist in planning and managing artificial recharge. Such a model is being developed by the CSIR.

PHASE 1: MINOR INFRASTRUCTURAL MODIFICATIONS

Total potential injection: 3 Mm^3/a

Main factors that affect start date:	Resolving "ownership" of injection water Erecting carbon filters & chlorinators Join the 500 mm Kleine Kuppe - Cimbebasia pipeline to the 300 mm Group 12 borehole pipeline, and to boreholes 9/6 and 9/9. Develop a permanent pumping system to Bh 12/3.
Available flow: (after modifications)	72 m^3/hr to Bh 9/8A 80 m^3/hr to Bh 9/14 >60 m^3/hr to Bh 9/6 >60 m^3/hr to Bh 9/9 110 m^3/hr to Bh 12/3 Total: ± 380 m^3/hr to all injection boreholes

The injection scenario is outlined in Table 8.

Table 8 Phase 1 injection scenario

Borehole Number	*Estimated Injection Rate (m^3/hr)*	*Estimated Monthly Injection Volume (m^3)*	*Estimated Yearly Injection Volume (m^3)*
Bh 9/8A	60	43000	520000
Bh 9/6*	60	43000	520000
Bh 9/9*	60	43000	520000
Bh 9/14*	60	43000	520000
Bh 12/3**	110	79000	960000
Total estimated injection	± 350	± 250000	± 3000000

* Injection step tests and short duration constant rate tests still need to be done in order to establish production injection rates.

** 190 m^3/hr will be available for injection at Bh 12/3 if there is no injection in boreholes 9/14.

Main pro: Relatively cheap and easy to implement.
This should reduce the "recovery" of the water level depression around borehole 9/8A from about 6 years to less than 2 years.

Main con: Most of the injection is not in the prime injection area (in the southern-most part of the aquifer).

PHASE 2: UP-GRADE TO A FULL-SCALE BOREHOLE INJECTION SCHEME

Total potential injection: 5.5 Mm^3/a

Main advantage over Phase 1:	Injection of 3.5 Mm^3/a into Group 12 area, as opposed to 1 m^3/a into this area.
Main factors that affect start date:	Same as phase 1, plus Raise funds for, and construct a full-scale injection scheme in the Group 12 area.

This is the preferred option, and is limited by finances. This option entails utilising the Group 12 boreholes for large-scale injection; drilling new injection boreholes south of the Gp12 boreholes; and installing infrastructure to supply them with treated water. The injection scenario is outlined in Table 9.

Table 9 Phase 2 injection scenario

Borehole Number	*Estimated Injection Rate (m^3/hr)*	*Estimated Monthly Injection Volume (m^3)*	*Estimated Yearly Injection Volume (m^3)*
Bh 9/8A	60	43000	520000
Bh 9/6	60	43000	520000
Bh 9/9	60	43000	520000
Bh 9/14	60	43000	520000
Bh 12/3	110	79000	960000
Bh 12/1A	100	72000	876000
New borehole 1* (possibly at 9/17)	100	72000	876000
New borehole 2 (possibly E of 12/1 & S of 12/4)	100	72000	876000
Total estimated injection	± 600	± 460 000	± 5 500 000

* This area has potential for a number of injection boreholes, and is discussed under Phase 3.

Main pro: High injection rates in the optimum part of the aquifer.
This should reduce the "recovery" of the water level depression around borehole 9/8A from about 6 years to less than 2 years, **and** top-up the main storage part of the aquifer by about 3.5 Mm^3/a.

Main con: High capital costs. Prior to implementing a full-scale artificial recharge scheme, a cost-benefit analysis should be done which compares various water supply options.

PHASE 3: ASSESS OTHER AREAS WITHIN THE WINDHOEK AQUIFER THAT MAY BE SUITABLE FOR ARTIFICIAL RECHARGE

Both the potential for borehole injection and sand dams in other parts of the Windhoek aquifer should be investigated. The approach to borehole injection tests should follow the procedures that have been done on Group 9 and 12 boreholes. The potential artificial recharge areas outlined below, are listed in order of preference.

East and west of the 12/5 and 12/6 boreholes
An area where artificial recharge could conceivably be of great value is the area east and west of the 12/5 and 12/6 boreholes. Considering that the Auas quartzites are believed to form the main storage area of the Windhoek aquifer, it would make sense to focus on recharging them wherever it is logistically possible. The area south west of borehole 12/6 could be one such area. The water levels in boreholes 12/5 and 12/6 have steadily dropped by 10 m since the early 1980's. This may well be due to high abstraction south of this area, and not as a result of abstraction from these two boreholes. This area is however outside (east) of the Windhoek graben, and it is not known how well this area is linked to the rest of the Group 12 and Group 9 wellfields. In theory, the same principal applied to recharging the Group 12 boreholes in the 12/1 - 12/3 area could apply here: artificially recharge the main storage area in order to ensure long-term sustainability of the aquifer.

West of borehole 12/1
An area that is included in Phase 2 recommendations is west of borehole 12/1. This area is believed to be highly fractured and coincides with the change in strike of the Auas quartzites from a ENE-WSW orientation to a NE-SW orientation. The prominent Venusberg fault forms the western boundary of this potential artificial recharge area. The reason for mentioning this area again, is because it may have the potential (due to its high expected permeability) to receive far more water than the 100 m^3/hr suggested in Phase 2. It may be possible to install a number of injection boreholes in this area and increase the artificial recharge rate considerably.

Group 7, the southernmost parts of the Group 6, and Group 11 wellfields
From historical water level data, it appears as if the Group 7 wellfield and possibly the southernmost boreholes of the Group 6 wellfield may be suited for artificial recharge. Water levels in the Group 7 wellfield have dropped by 50 m over the past 50 years, and during periods of high abstraction, the water levels dropped almost 90 m below their 1950 levels. The water levels in the southernmost parts of the Group 6 wellfield (boreholes 6/9 and 6/10) have not dropped much since the late 1960's, however during periods of high abstraction, their water levels tend to drop by up to 20 m. Like the Group 9 wellfield, borehole injection in these areas should be considered as a means of rapidly replenishing the storage space created after periods of high abstraction.

Possibly a better option than artificially recharging the Group 6 or 7 wellfields would be to recharge the Group 11 wellfield. This area is hydraulically up-gradient of the Group 6 and 7 wellfields, and assuming the area is sufficiently permeable, and has the available space, then the same principal that is proposed for the Group 9 and 12 areas could be applied. This is to recharge the up-gradient area and replenish the aquifer which has been drained as a result of extended, large-scale abstraction, rather than recharge localised depressions that have formed in the wellfields.

Sand dams

Favourable sites where permeable fracture zones cross river beds should be identified as possible sand dam sites, and infiltration tests should be done at these sites. The aim of the sand dams should be to lengthen the period of infiltration to the hard-rock aquifer. Unlike conventional sand dams, these dams should be located in areas where they will rapidly lose their water through infiltration to the aquifer below.

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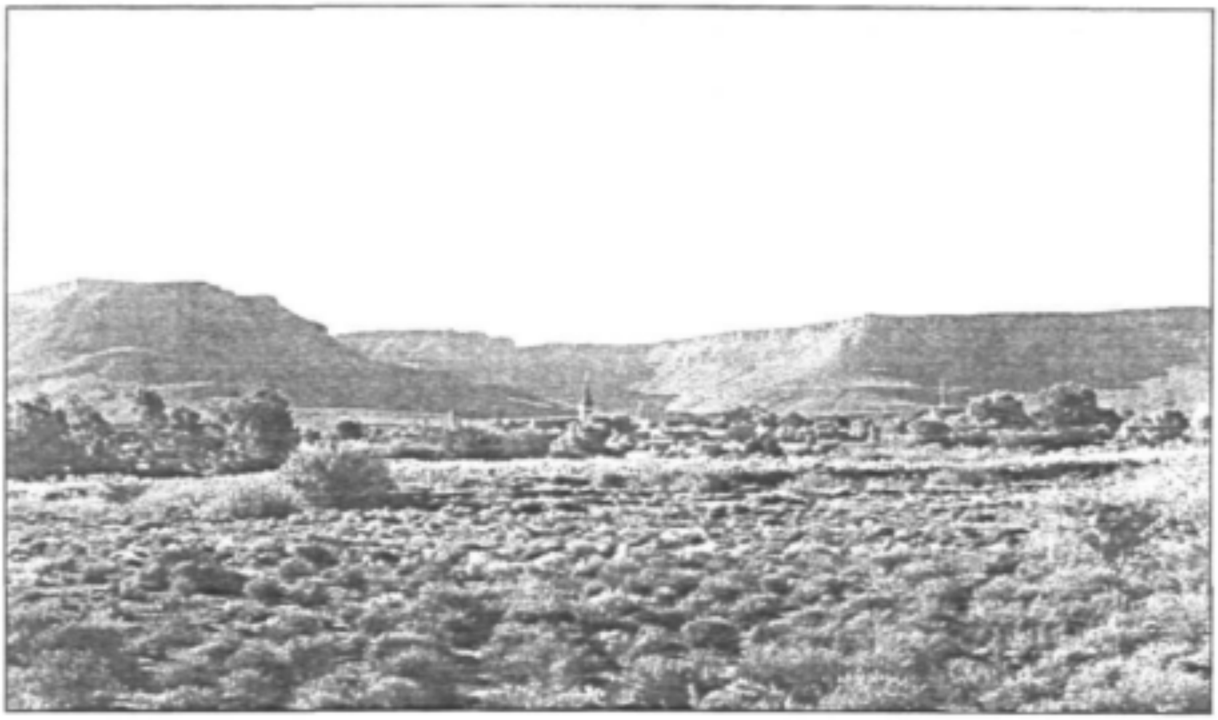
SECTION 2:

Calvinia Pilot Artificial Recharge Study

by

L Cavé, E C Murray & G Tredoux



Calvinia with the Hamtam Mountains in the background

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1. INTRODUCTION

The town of Calvinia, on the western edge of the Karoo, has a mean annual precipitation of 210 mm and a long history of water shortages. Artificial groundwater recharge was proposed as a means of augmenting the scarce water supplies of this town. Most of the town's water comes from the Karee Dam which is located above the town in the foothills of the Hantam Mountains. A small proportion of the town's water is derived from groundwater, which, to a large extent, is saline and has to be blended with surface water in order for it to be potable. The artificial recharge scheme involves storing surface water in a highly brecciated geologic structure called a plug or pipe. This water would serve as a reserve to be used during droughts or periods of high demand.

Scattered throughout the Karoo and Namaqualand, are a number of breccia pipes. These pipes were formed as a result of hydrothermal explosions, and are thus highly permeable. Borehole yields are commonly in excess of 50 L/s. The pipes are not used as a source of sustainable groundwater, because natural recharge is too low. Once the Calvinia pipe, known as the Kopoasfontein breccia pipe, was dewatered up to a depth of 126 m, it took three years for the water level to recover.

The Kopoasfontein breccia pipe, a roughly cylindrical body of fractured rock, 12 km east of the town, is located in Permian mudstones, shales and siltstones, and is intensely fractured and brecciated to a depth of 250 m. Its full storage potential is not known. It was initially estimated to hold 57 000 - 74 000 m^3 (Murray and Tredoux, 1998). Its usable storage however depends on the depth of the column of the pipe that is utilised. More recent estimates of the available water to a depth of 182 m, which is where the porosity starts to decrease, is 80 000 m^3, and up to a depth of 255 m, after which the porosity becomes very low, is 105 000 m^3.

The town's maximum winter consumption is 1 100 m^3/day, and their maximum summer consumption is 1 600 m^3/day. The reserves in the breccia pipe are thus equivalent to two to three months supply for the town (using maximum consumption figures), and is considered a valuable back-up to their vulnerable surface water source.

The existing municipal pump can unfortunately only draw the water level down to 142 m. The abstractable volume up to this level is 48 000 m^3. If the municipality had a pump that could draw the water level down another 40 m, they could add another month's back-up, or more, to their water bank.

In order to take advantage of this sub-surface reservoir, it will need to be artificially recharged after each time it has been drained. The source of the water used for artificial recharge is treated surface water from the Karee Dam, which is injected via boreholes into the breccia pipe.

The residence time of injected water is affected by the unusual geochemistry of the breccia pipe system. At the time of writing this report only one injection run had been completed. In order to sufficiently dilute the breccia's water so that it is fit for human consumption, a few injection runs will be needed.

Because of the complex geochemistry of the breccia pipe, this project has focussed primarily on understanding the water-water and water-rock interactions that will take place once fresh water has been injected into the pipe.

2. GEOLOGY

Calvinia is situated within the Karoo basin, with sediments of the Dwyka Formation and Ecca Group forming the dominant surface geology (Figures 1 & 2). The sediments have been intruded by a profusion of dolerite dykes and sills (Walker and Poldervaart, 1941), as well as by isolated breccia pipes (Hallbauer *et al.*, 1995).

The Permian Ecca Group generally consists of a series of medium to dark grey mudstones. These are represented by the Tierberg, White Hill and Prince Albert Formations in the Calvinia district. The White Hill Formation is characterised by black, carbonaceous shales that are known for their occasional sulphide mineralisation (Fe, Zn) and have been unsuccessfully prospected for oil reserves (Cole and McLachlan, 1990). The Prince Albert Formation is underlain by the glacial sediments of the Dwyka Formation (Late Carboniferous, Early Permian), consisting of bedded diamictites and mudrock, containing ice-rafted remnants.

2.1 Dolerite dykes and sills

Dolerites, mostly olivine tholeiites of Jurassic age, form a complex network of dykes (up to 100 km long) and sills (up to 100 m thick) over many areas of the Karoo. The thickest of these have intruded along preferential horizons such as the contact between the Dwyka and Prince Albert Formations. Sediments at the contact with the dolerite sills are usually altered to hornfels or even metasomatised into a granophyre-type rock (Walker and Poldervaart, 1941).

2.2 Breccia pipes

Several breccia pipes occur in the area around the town of Calvinia (Hallbauer *et al.*, 1995). These appear as circular features on the surface which can be identified by their negative, shallow topographic relief or positive topographic relief on aerial photographs, or by magnetic anomalies. The pipes are usually between 50 and 100 m in diameter and vary in geological appearance and mineralogical composition. They are expected to be roughly cylindrical in shape below the surface. Hallbauer *et al.* (1995) recognised two facies of breccia pipe in the Calvinia area, based on textural characteristics:

- domed, baked, molten, recrystallised and contorted sediments containing clasts from other underlying strata; and
- true breccia with broken, displaced and recemented blocks.

The Kopoasfontein breccia pipe

At Kopoasfontein, approximately 12 km east of Calvinia, a 755 m core was drilled into a breccia pipe. Figure 43 shows the different facies recognised from the geological core, as logged by Chevallier and Balaban (1995). A second borehole, drilled into unbrecciated sediments at the same location, shows the corresponding Ecca and Dwyka formations.

The Kopoasfontein breccia pipe is approximately 100 m in diameter and produces a strong magnetic anomaly. At surface the sediment looks broken and re-cemented, but still homogeneous in composition. Both Ecca and Dwyka groups can be recognised in the core but their boundary is very uncertain. A detailed stratigraphy of the Ecca is also impossible to establish. The formations have been roughly subdivided in Figure 6, based on the positions of dolerite sills.

The fractures and cavities in the breccia pipe are partially filled by hydrothermal minerals, which were transported by external fluids. Sulphide minerals (pyrite, pyrrhotite, chalcopyrite, sphalerite, galena, marcassite, bornite and covellite), quartz, calcite, siderite, gypsum, barite, chlorite, fluorite and apophyllite have been observed, although a paragenetic sequence for the mineralisation has not yet been defined. Sulphide mineralisation (pyrite, chalcopyrite and sphalerite) was also recognised in borehole chippings from the breccia pipe on the farm Witwal, just east of the town.

The origin of the brecciation, and subsequently the mineralisation, appears to be linked to two early dolerite sills which intruded at the Dwyka-Prince Albert and Prince Albert-White Hill contacts. Alteration features along the sediment-dolerite interface include localised magmatic differentiation and layering in the dolerite, extraction of the leucocratic phase, injection of material into the sediments and digestion of sediment material. Hallbauer *et al.* (1995) thus concluded that local hydrothermal, contact metasomatic reactions had mobilised elements from the White Hill and Prince Albert Formations. Fluids were then transported along the pressure gradient that formed between the intrusive contact and the brecciated rock, and the secondary minerals deposited in the low-pressure/low-temperature environment of the brecciated column.

Two later dolerite sills of minor importance have been intruded higher up, at approximately 27 m below surface in the Tierberg Formation and near the Tierberg-White Hill contact at 380 m. These intruded after the brecciation and the mineralisation took place. The breccia pipe is, therefore, supposed to be early dolerite in age (Jurassic).

Borehole geological logs are provided in Appendix 2.1.

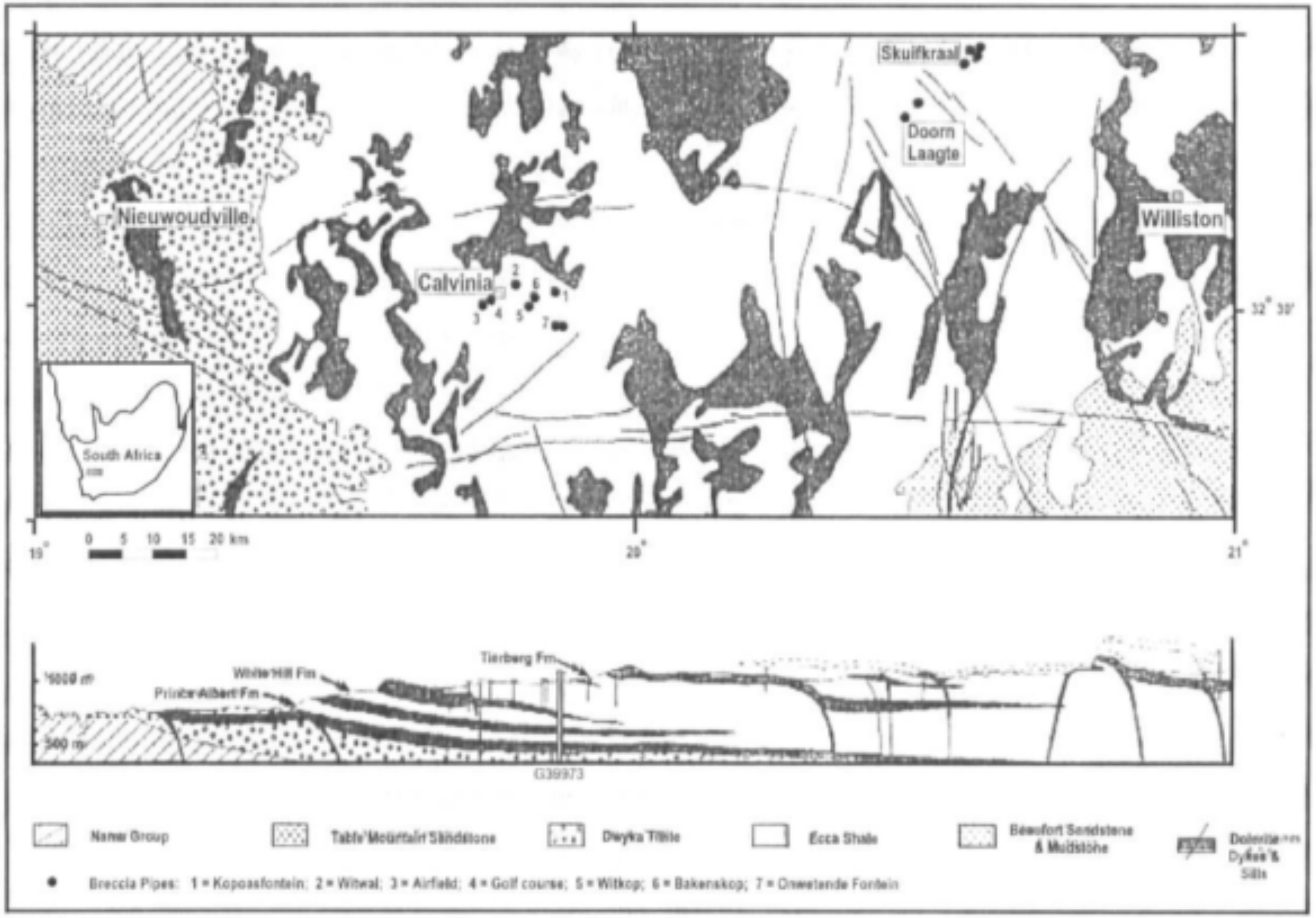

Figure 1 Geological map of the Calvinia area, showing the dominant stratigraphic formations and the position of breccia pipes (after Hallbauer *et al.*, 1995).

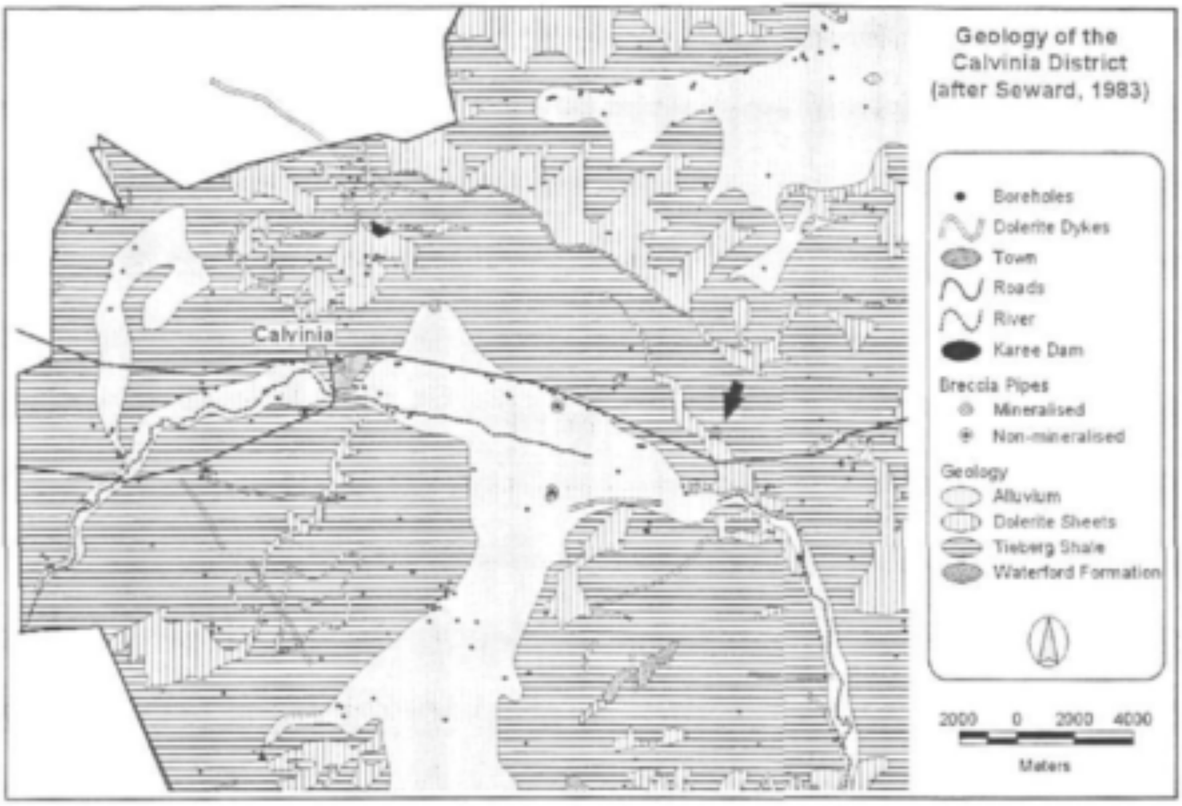

Figure 2 Location of Kopoasfontein (arrow) and other breccia pipes around Calvinia after Seaward, 1983)

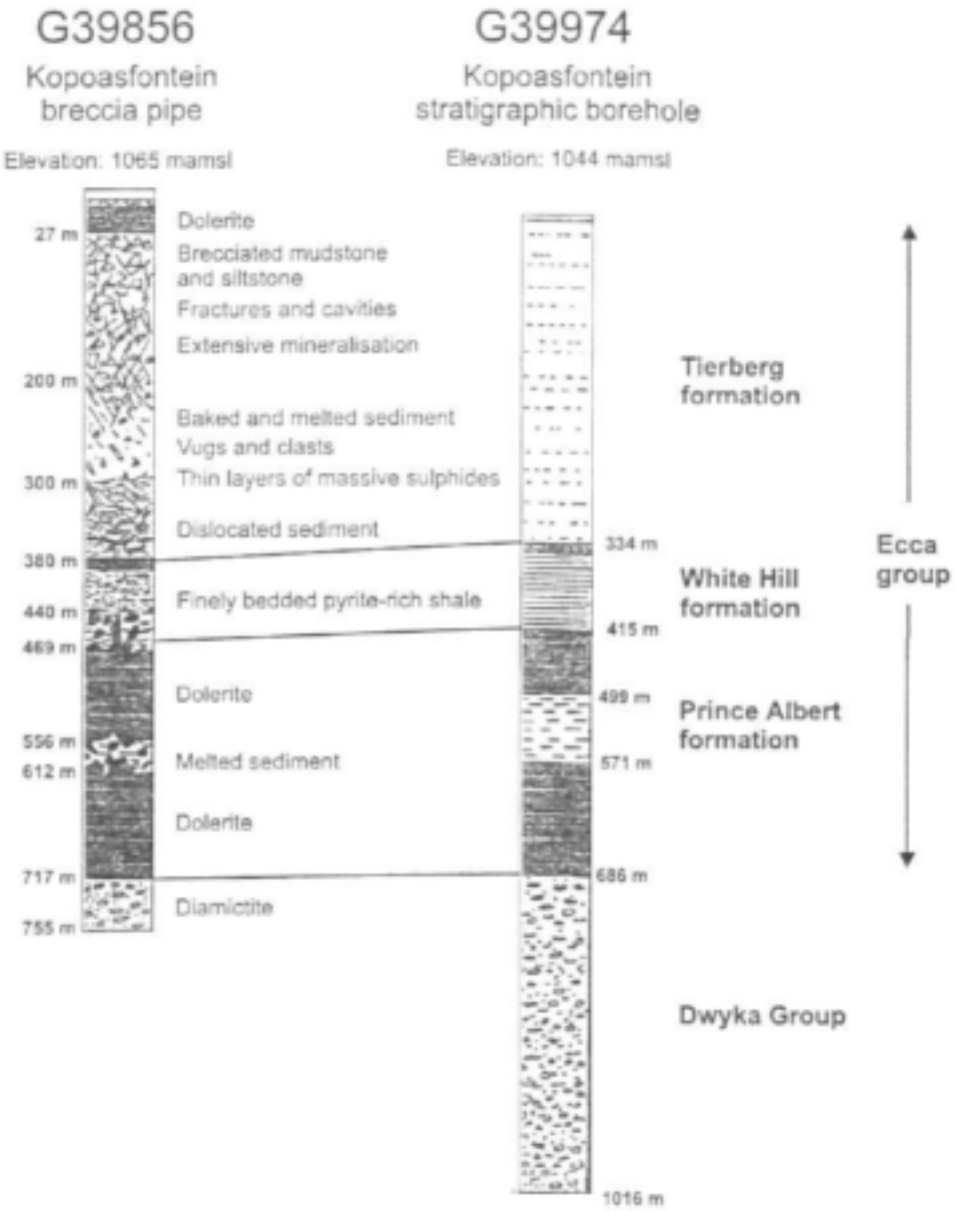

Figure 3 Lithological sequence of boreholes G39856 and G39974 at Kopoasfontein, as logged by Chevallier and Balaban (1995).

3. THE HYDRAULIC SUITABILITY OF THE KOPOASFONTEIN BRECCIA PIPE AS A SUB-SURFACE STORAGE RESERVOIR

3.1 Results for the first pumping test programme

A number of boreholes, including a 755 m deep core borehole, were drilled in and around the Kopoasfontein breccia pipe (Figure 4).

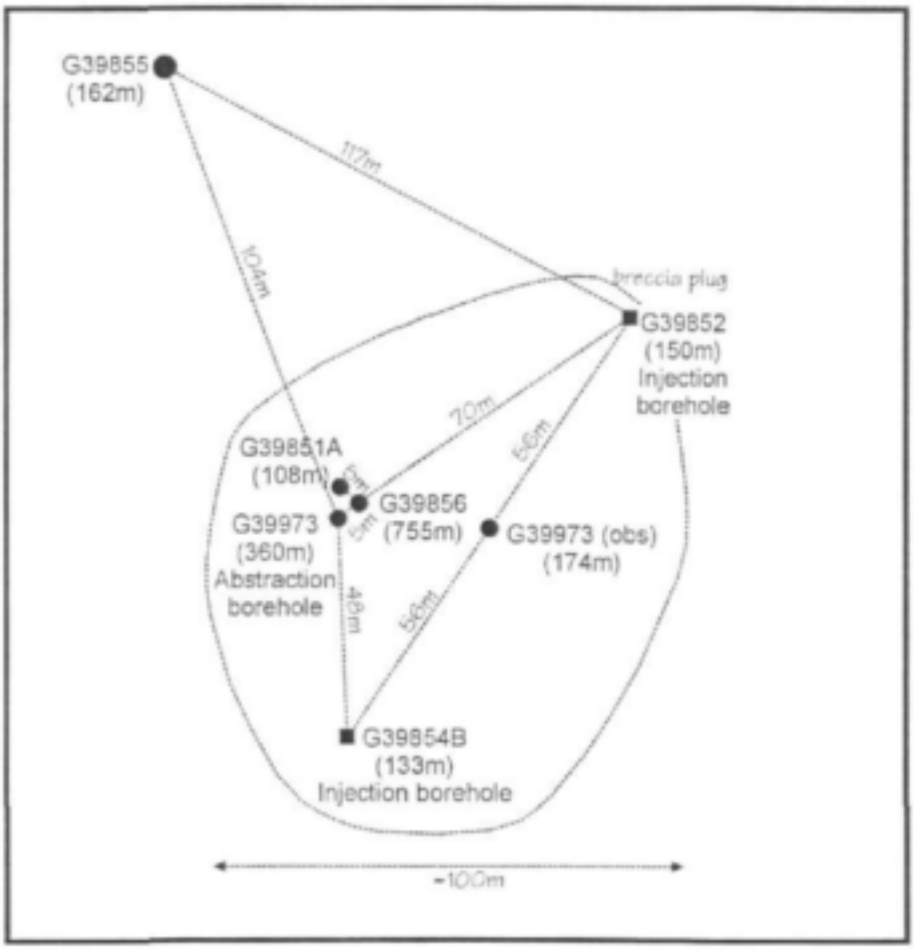

Figure 4 Calvinia's artificial recharge site

Borehole G39973 (Plate 1), which is 360 m deep, was test pumped by DWAF in 1994 (Woodford, 1994). After the drilling and a multiple rate (step) drawdown test, the water level in the borehole did not recover prior to the constant discharge test. Although the rest water level in the borehole is around 19 mbgl, the constant discharge test was started with the water level at 73.8 mbgl. After 7 days and 19 hours of pumping at 24.4 L/s, the water level in the borehole dropped to 126.2 mbgl; and after 19 days of recovery, the water level had only recovered to 120.5 mbgl (Figure 5). This showed that this borehole does not penetrate a dynamic groundwater system, but rather, a groundwater compartment that once pumped, will take a long time for the water level to recover to its original position.

The test also showed that boreholes which also penetrate the pipe, like G39852 and G39854, responded in the same manner as the pumping borehole; and boreholes outside the pipe, like borehole G39855, barely responded at all. This showed that the pipe behaves, hydraulically, as a single unit, and that any borehole outside of it belongs to a separate groundwater system which is poorly linked to the pipe.

Plate 1 Abstraction borehole G39973

During the step drawdown and constant discharge tests, a total of 28 800 m^3 was abstracted, and the water level was drawn down to 126 mbgl. The water level took 1128 days to recover to its pre-pumping water level (Figure 6). This implies that the rate of leakage into this groundwater compartment from the surrounding Karoo formation is 25 m^3/day. This is the approximate sustainable yield of the compartment without artificial recharge.

If the pipe is artificially recharged, it would be possible to abstract water at the town's daily required rate of about 1000 m^3/day (or 12 L/s) for lengthy periods. The permeability of the pipe is such that water can be abstracted at a far greater rate than this, thus should the town require this water to cover peak demand periods, or more than 1000 m^3/day, the pipe could be pumped at 24 L/s or more. The limiting factor is more likely to be the diameter of the supply pipeline than the pipe's yield.

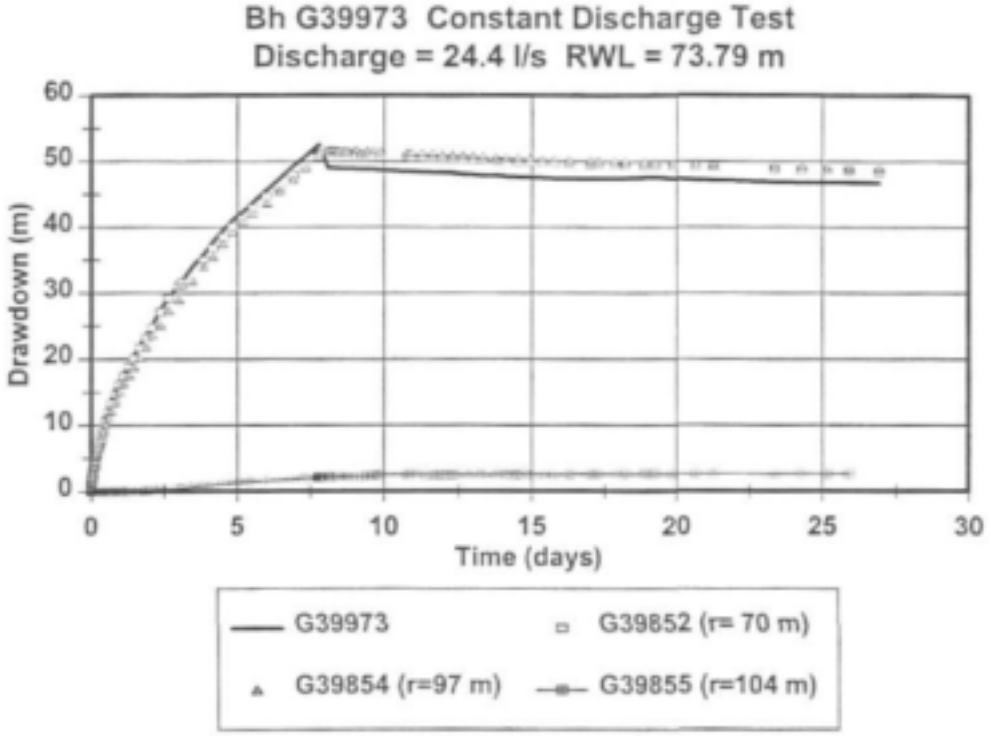

Figure 5 Constant discharge test on borehole G39973

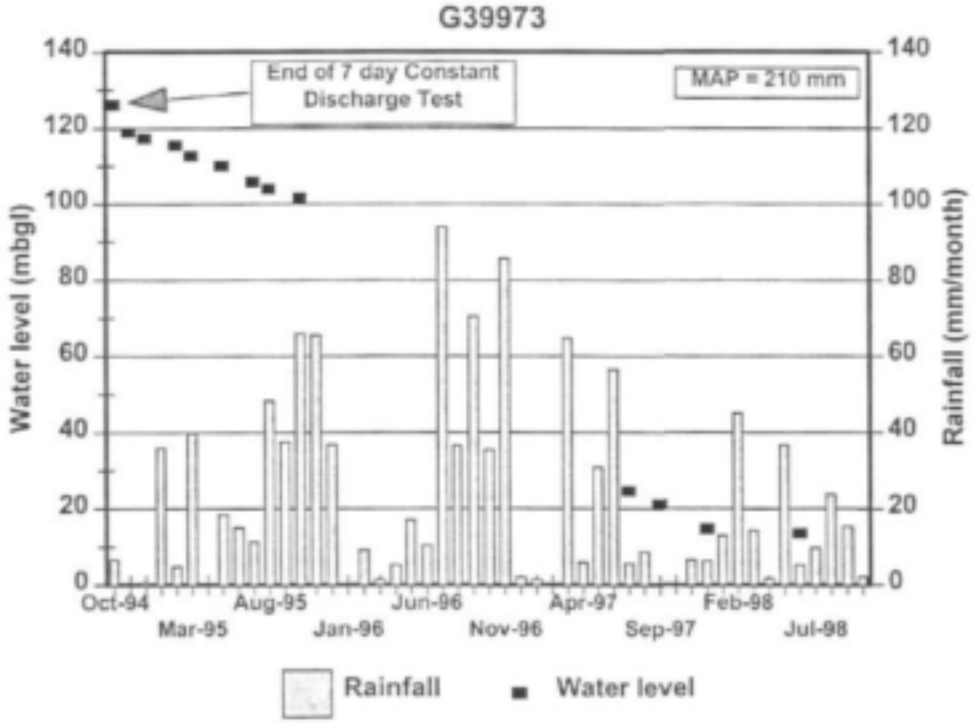

Figure 6 Water level recovery in borehole G39973 after the constant discharge test

3.2 *The storage capacity of the pipe*

The full storage potential of the pipe, or "sub-surface reservoir" as it is commonly called, is not known. It was initially estimated to hold 57 000 - 74 000 m^3 (Murray and Tredoux, 1998). Its usable storage however depends on the depth of the column of the pipe that is utilised. In order to obtain a better estimate of its storage potential, borehole G39973 was geophysically logged by DWAF (Figure 7). This provided an indication of porosity at depths greater than 126 mbgl, which was the water level at the time of the logging. The results of this showed that at borehole G39973, the zone of highest porosity is up to a depth of 182 m, and that below 255 m, the porosity drops off considerably.

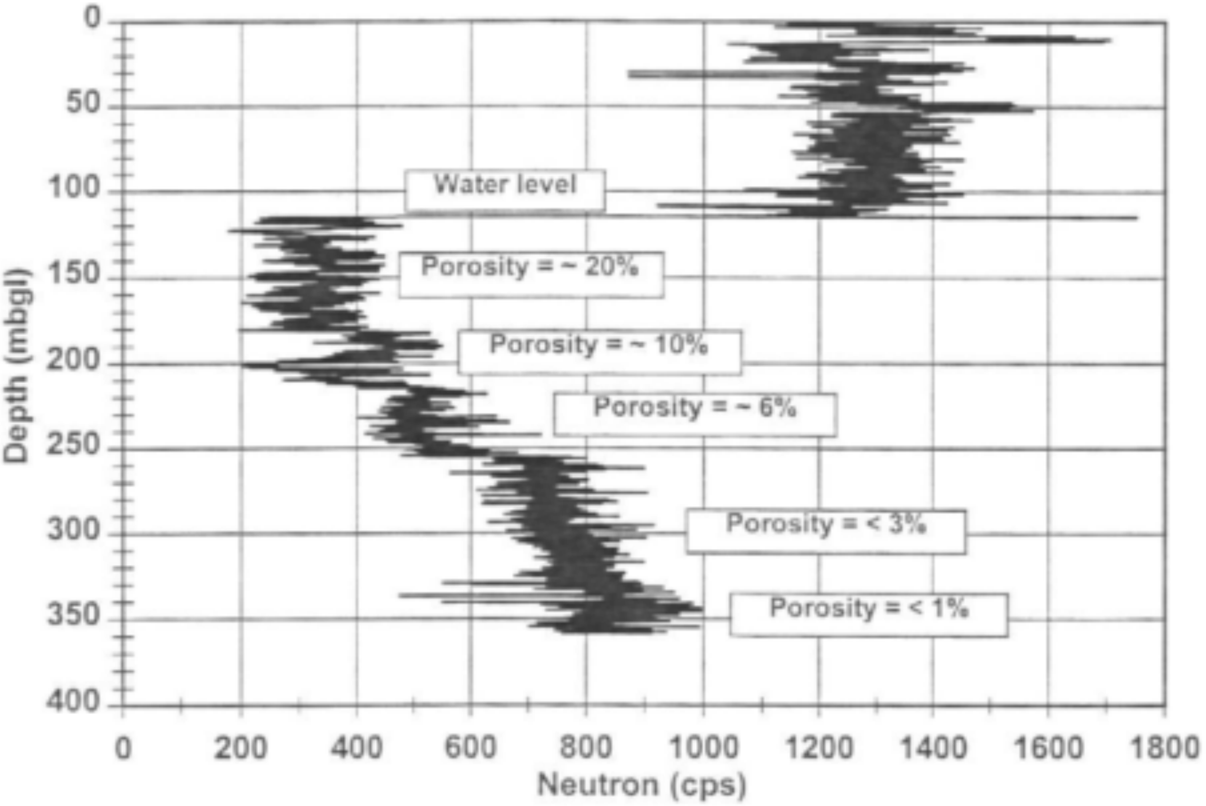

Figure 7 Neutron log of borehole G39973 with estimated porosity values

Prior to injecting treated dam water into the pipe, the existing, poor quality water had to be removed. This was done between June 1999 and April 2001. It took longer than expected since borehole G39002 had to be unblocked and a pump powerful enough to abstract from 255 m needed to be obtained. At first, the municipality's pump was installed at 125 mbgl and the water was pumped to a depth of 115 mbgl. DWAF unblocked the blockage at 127 m, and the municipal pump was installed at 180 mbgl after an unsuccessful attempt to gain use of a DWAF pump which was thought (incorrectly) to be able to pump a head of 255 m.

By April 2001, 48 517 m^3 of water had been removed from storage, and the maximum water level that the municipal pump could draw water down to, was 142.6 mbgl. Figure 8 provides an estimate of how storage increases with depth up to 140 m based on abstraction and injection data; and an estimate of how storage increases with depth between 140 m and 255m based on porosity estimates from geophysical data.

The volume of water pumped between 120 m and 142 m was 18 096 m^3, or an average of 822 m^3/m. If the porosity at borehole G39973 is representative of the entire pipe, then this value is likely to be the maximum for the pipe; and it is likely to decrease significantly below 182 m, and then again below 255 m. This figure may be slightly exaggerated, since during the drilling of G39973(obs), a water strike at 12 mbgl, at the top contact of the dolerite sill, was struck, and this led to perched water leaking into the pipe. The rate of long term leakage is not known, however, at the time of drilling, the water strike gave a blow yield of 4 L/s.

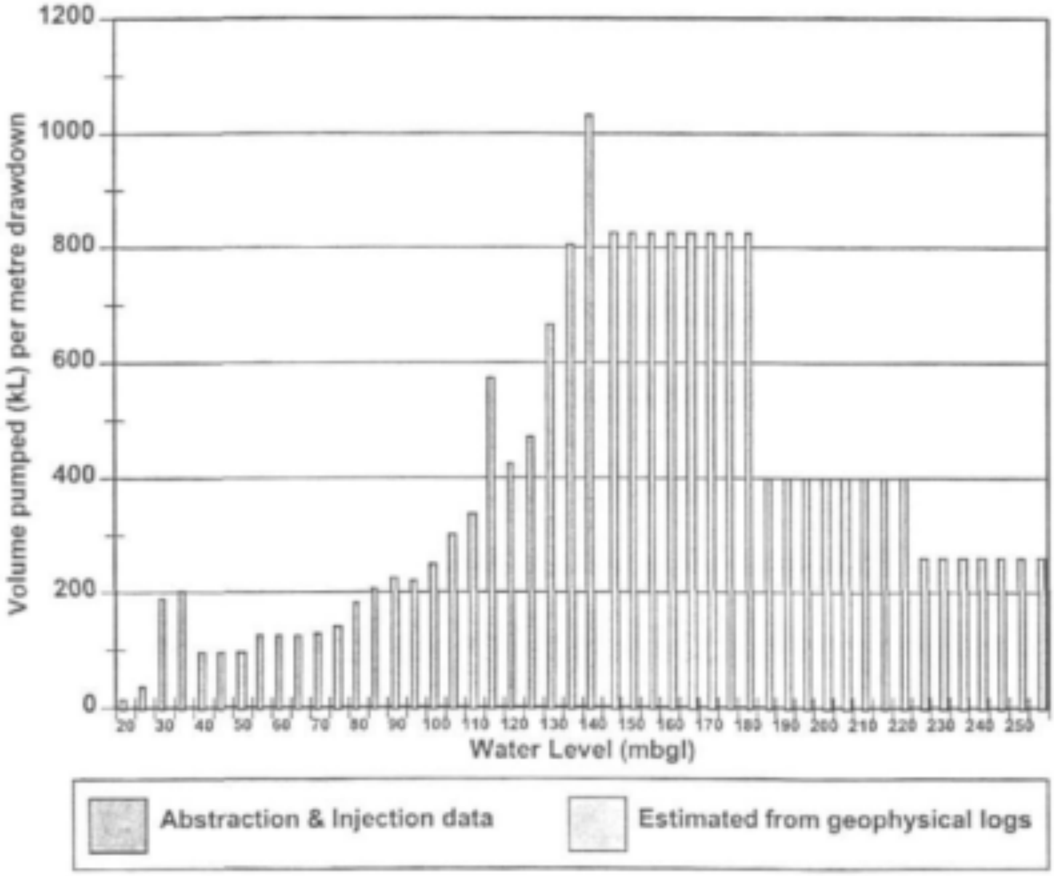

Figure 8 Estimated change in storage with depth in the breccia pipe

Table 1 gives revised abstraction volumes based on the pumping data up to a depth of 142 m, and estimated porosity values below this depth at borehole G39973. Possibly the most reasonable estimate of the pipe's available water resource is 81 000 m^3 up to a depth of 182 m and 106 000 m^3 up to a depth of 255 m.

An unknown factor is whether the pipe is linked to a deep groundwater system. If this was the case, the piezometric level would have to be deeper than 142 m. The only way to establish this would be to draw the water level down as far as possible, and observe the recovery. If the pipe is indeed linked to a deeper groundwater system, then the water level will recover fairly quickly to the piezometric level, and after that, it will recover very slowly, like it did after the pumping tests data presented earlier. The implications of the pipe being linked to a deeper groundwater system is that its sustainable yield would be far greater, because of the vast recharge area that would feed the deeper groundwater system. This however will remain an unknown factor until water is removed from great depths within the pipe

Table 1 Estimated abstractable volume of the subsurface reservoir

Method 1: Using the known abstractable volume and porosity only

Depth (mbgl)	*Volume (m^3)*	*Comments*
19 - 142	48 000	known volume
142 - 182	33 000	porosity = 20%
Total up to 182 m	**81 000**	
182 - 220	16 000	porosity = 10%
220 - 255	9 000	porosity = 6%
Total up to 255 m	**106 000**	

Method 2: Using known volume, porosity and estimates of the pipe's radius

Depth (mbgl)	*Volume (m^3)*	*Volume (m^3)*	*Volume (m^3)*	*Comments*
	Radius of pipe = 25 m	Radius of pipe = 50 m	Radius of pipe = 75 m	
19 - 142	48 000	48 000	48 000	known volume
142 - 182	16 000	63 000	141 000	porosity = 20%
Total up to 182 m	**64 000**	**111 000**	**189 000**	
182 - 220	7 000	30 000	67 000	porosity = 10%
220 - 255	4 000	16 000	37 000	porosity = 6%
Total up to 255 m	**75 000**	**157 000**	**293 000**	

4. THE INJECTION AND MONITORING BOREHOLES

4.1 The injection boreholes

There are two injection boreholes, G 39852 and G 39854B (Figure 1 and Plates 2).The Department of Water Affairs and Forestry (DWAF) drilled a new injection borehole, G39854B, after attempts to case the existing injection borehole, G39854, failed. The new injection borehole is 133 m deep, and it penetrated the brecciated material. The drilling "blow" yield was estimated to be more than 50 L/s.

This borehole has 203 mm poorly slotted casing from 0 - 68.2 m; factory slotted casing for injection from 68.2 - 103.5 m; and poorly slotted casing for 103.5 - 110.8 m. Thereafter, the borehole is open. The factory slotted casing consists of 300 mm by 10 mm slots throughout the length of the casing, and will allow for easy flow of water from the borehole into the breccia pipe. The borehole should have been cased with this casing up to its completion depth of 133 m, but this proved to be problematic. The reason for this may be because the hole is skew as a result of intersecting cavities during drilling.

Borehole G39852 was left uncased because borehole G39854B intercepted highly permeable parts of the breccia pipe. Considering that the injection rate is only about 10 L/s due to the restricted diameter of the supply pipeline, only one injection borehole (G39854B) is needed, even though both boreholes are equipped for injection.

Plate 2 Injection borehole G39852 in the foreground and abstraction borehole G39973 in the background

4.2 The monitoring boreholes

There are two monitoring boreholes that are of particular value to this project: G39973(obs) (Plate 3) which was drilled in 1999 by DWAF, and which penetrates the breccia, and G39855, which is outside the breccia. Table 8 gives the borehole specifics.

Table 2 Monitoring boreholes

Bh No	*Depth (m)*	*Diameter*	*Casing*	*Comments*
G 39973(obs)	174	0-174m: 8"	0-12 m: 6" plain steel 0-174 m: 65 mm ID, slotted every 6 m	This borehole was drilled to take a data logger or a dip meter probe. Old drill rods were used as the casing (in order to cut costs)
G 39855	162	8"	6"	Outside the breccia
G 39851A	108	8"	6"	Inside the breccia
G39856	755	±60 mm	±60 mm	This core borehole is inside the breccia, but sealed and unusable.

Plate 3 Drilling the monitoring borehole G39973(obs)

5. GEOCHEMISTRY

5.1 Introduction

Prior to utilising the breccia pipe for sub-surface storage, a comprehensive water quality assessment was undertaken. This assessment formed Ms L Cavé's M.Sc thesis, and a summary of this work is included in this chapter. Two pertinent questions were addressed:

Which water source should be used for sub-surface storage, the treated dam water or the natural Karoo groundwater? The reason for considering Karoo groundwater, which is very saline, was because the municipality had already equipped the boreholes, and it was thought that by blending this water with the breccia water, the resultant water may be potable. The breccia water, although not highly saline (in terms of electrical conductivity), is itself unfit for human consumption because of the high concentrations of certain ions, and needs to be blended in order to make it potable.

The second question was what would the blended water quality look like? The feasibility of this scheme depends on whether the water held in storage within the breccia pipe is potable, and how long it takes for the water to become unfit for human consumption. If water can be stored for months or years in the breccia pipe before it becomes undrinkable, then the scheme would be successful, however, if the water rapidly "picks up" the undesirable ions held in the breccia water, then the scheme would have to be abandoned or only used when blended with surface water.

The ambient water quality in this geologic structure is unsuitable for human consumption, primarily because of the high fluoride and arsenic concentrations. If this water were to be used for domestic consumption over short periods, it would be necessary to dilute it with dam water at a ratio of at least 1:8 (ie. one part pipe water to eight parts dam water). The letter from the CSIR to the then Town Engineer, Mr EJ Wentzel, recommending this mixing ratio and highlighting the health-risk implications is attached in Appendix 2.2

Selected extracts from Ms L Cavé's M.Sc thesis and a subsequent paper published in the proceedings of the IAH Congress (Cavé, 2000) have been incorporated in the following sections.

5.2 Geochemical factors affecting artificial recharge

Introducing recharge water with different chemical properties from that of the *in situ* groundwaters may shift chemical equilibria in the recharge and native waters and affect their chemical interaction with aquifer solids. Changes in the physical and chemical properties of the system, such as temperature, pressure, gas concentrations, pH and redox potential may also enhance or inhibit reactions, or cause one reaction to be favoured over another (Pyne 1995). Effects can range from mineral precipitation or dissolution or gas generation to immobilization/mobilisation of unwanted chemicals by sorption to or desorption from aquifer surfaces.

Two major issues that generally arise from chemical reactions are changes in the permeability of the system and changes in the quality of the recovered water. Clogging of borehole screens or the aquifer itself (from mechanical blockage by suspended solids, biological activity, gas entrainment or mineral precipitation) is a major problem affecting borehole injection. Injection of turbid water, mobilisation of clay particles, nucleation of precipitates and dissolution of cementing materials could cause clogging by suspended solids. Typical precipitation problems involve carbonates or Fe or Mn oxyhydroxides, but any sparingly soluble phase may precipitate under favourable conditions.

Changes in water quality can be beneficial, as in the case of soil-aquifer treatment of wastewater or dilution of unwanted groundwater constituents to acceptable levels using high quality recharge water. Artificial recharge could, however, mobilise contaminants such as Rn, As, F, Fe, Mn, NH_4 and organic compounds from natural sources in the aquifer, degrading the recovered water. Residence time is an important factor in determining the extent of accumulation.

5.3 Geochemistry of water sources and the rocks of the breccia pipe

The compatibility of water sources from Calvinia and rocks from the Kopoasfontein breccia pipe, together with the geochemical processes governing their interactions, may all play a role in determining the success of the artificial groundwater recharge scheme.

Chemical analysis of samples from the various water sources and mineralogical and bulk chemical analysis of rock samples from a drilling core were conducted. This allowed the various chemical systems to be delineated, so that predictions could be made regarding the fate of species during mixing and water-rock interaction. Geochemical information of this kind is also necessary for the successful management of the scheme.

5.3.1 Composition of water sources

Chemical characteristics of the recharge sources (Karee Dam, G39638, G39648, G39627A and G39861) are compared with those of the native groundwater (G39973) in Table 3. Major ions were analysed by ion chromatography, molybdate-reactive silica by colorimetry, total iron by flame atomic absorption spectrophotometry and arsenic by inductively coupled plasma-mass spectroscopy.

Table 3 Chemical composition of water sources in the Calvinia borehole injection scheme (data in mg/L unless other units specified)

	Potential recharge sources					
	Dam	*Karoo boreholes*				*Breccia*
Parameter		*G39648*	*G39638*	*G39627A*	*G39861*	*G39973*
Field pH	7.1	8.1	7.8	7.7	8.6	9.8
EC (mS/m)	19	435	571	850	73	71
T.Alk (meq/L)	1.20	5.21	7.54	8.00	4.03	3.06
Na	13	590	645	1270	180	170
K	0.8	5.8	3.9	5.3	0.7	1.7
Ca	17	220	340	260	4.6	5.3
Mg	12	210	340	600	2.0	2.6
NH_4	<0.1	<0.1	<0.1	<0.1	<0.1	1.3
Cl	16	1570	1970	2585	150	120
SO_4	20	150	480	1200	5.7	54
NO_3	0.4	<0.1	<0.1	<0.1	<0.1	<0.1
F	0.1	1.1	0.4	0.8	2.7	12
Si	3.7	5.0	7.7	7.0	5.7	8.6
Total Fe	0.08	0.07	0.13	0.10	0.05	0.36
As (µg/L)	0.8	12	6.2	7.7	1.7	325

Note: Appendix 2.2 contains water quality guidelines to which these values can be compared

The Karee Dam stores runoff water from a nearby mountain catchment. This surface water is of very high quality, having low salinity and near-neutral pH. The dominant major ions in solution are Ca^{2+}, Mg^{2+} and HCO_3^-. Suspended particulates and dissolved organic carbon (6.6 mg/L) pose a risk of clogging or biofouling if surface water is used for artificial recharge without effective treatment and filtration. The water is oxygenated and may cause oxidation of reduced species in the breccia pipe. The pH buffering capacity (Alkalinity) of the surface waters is low and high pH is likely to result from reaction with the rocks and groundwater of Kopoasfontein.

The municipal boreholes, G39638, G39648 and G39627A abstract groundwater from shallow alluvial and weathered shale aquifers. These three waters are of similar composition, being hard, brackish and more reducing than the surface water. They are generally blended with the dam water when supplies are critically low, but undiluted they are not desirable for human consumption. The dominant major ions are Na^+, Mg^{2+} and Cl^-.

Groundwater from the Witwal breccia pipe (G39861) provides an alternative recharge source which is compatible with the Kopoasfontein groundwater, because of its similar geological origin. This water has a slightly lower pH and higher alkalinity than the Kopoasfontein groundwater, although both are of $Na!Cl/HCO_3/CO_3$ type. Witwal groundwater, however, also contains sufficient F^- to require treatment after recovery from the breccia pipe.

The natural breccia pipe water (G39973) has an unusually alkaline pH and low salinity. Species such as NH_4^+ and H_2S in solution indicate a reducing environment. High concentrations of F^- and As would pose a health risk for long-term consumers of the water and could become problematic if similar levels were to accumulate in the stored water. The geochemical environment in the breccia pipe (high pH, reducing conditions) favour mobilisation of F^- and $H_2AsO_3^-$ in the groundwater.

5.3.2 Breccia pipe composition and mineralogy

Major rock types in the upper portion of the breccia pipe are baked, fragmented mudstones and shales, which have been subjected to hydrothermal activity. This has caused some alteration of the original detrital minerals and precipitated new phases, especially along the cavities and fractures.

Rock mineralogy was determined by optical microscopy and X-ray powder diffraction analysis. The bulk argillaceous material comprises quartz, feldspar and clay minerals of the illite and chlorite groups. Late stage secondary minerals have also been deposited in the brecciated shale environment by hydrothermal solutions. Quartz, calcite, chlorite and metal sulphide minerals (pyrite, pyrrhotite, chalcopyrite, galena) occur as vein minerals in the breccia and as coatings on the walls of fractures and cavities. Well-formed crystals of fluorapophyllite, line many of the breccia cavities. Reaction with these minerals may have a strong influence on the groundwater composition in the breccia pipe, particularly since they are concentrated along zones of high permeability.

Bulk elemental composition was analysed by X-ray fluorescence spectrometry. Table 4 gives mean chemical concentrations based on the analysis of 14 core samples from the top 300 m of the breccia.

Table 4 Bulk composition of breccia pipe rocks

Major Oxides	*mean wt %*	*s.d.**	*Trace elements*	*mean mg/kg*	*s.d.**
SiO_2	60	3.4	F	360	320
Al_2O_3	18	1.7	As	11	6
Fe_2O_3	7.3	1.6	Ba	780	210
MgO	2.4	1.4	Mn	840	220
CaO	1.8	2.5	Cu	16	22
Na_2O	2.4	0.9	Pb	22	22
K_2O	3.7	1.1	Zn	53	25

*s.d. = standard deviation, n =14.

Base cations in the rock are dominated by K, Mg and Na, but Ca concentrations are generally low. High standard deviations for Ca, F and the trace metal concentrations reflect localisation of minerals containing these elements (such as calcite, apophyllite and metal sulphides) in the breccia pipe.

In the absence of measurable fluorite, dissolution of fluorapophyllite was proposed as a possible source of the high dissolved F^- in the groundwater. Other sources may be as a trace component in the clay minerals of the shales. Fluoride concentrations in solution are generally limited by the solubility of CaF_2, but low Ca^{2+} in the groundwater (due to low $CaCO_3$ solubility at high pH and possibly base cation exchange of Ca^{2+} for Na^+ on mineral surfaces) has allowed high F^- concentrations to accumulate. Alkaline conditions also favour the exchange of OH^- for F^- adsorbed to aquifer surfaces.

Metal sulphides in the Kopoasfontein breccia pipe, particularly pyrite containing traces of arsenic, are anticipated to be the source of dissolved As. Under reducing conditions As(III) forms arsenite oxyanions which have higher mobility than the oxidised arsenate forms, particularly at high pH when anion adsorption to mineral surfaces is low.

One of the considerations of the borehole injection scheme is the possibility of using recharge waters to manipulate the geochemical environment by lowering the pH and raising the redox potential. This could then contribute to immobilising F^- and As, in addition to any dilution effects, which might decrease their concentrations to acceptable levels.

5.4 Evaluation of potential geochemical reactions during artificial recharge

Quantitative geochemical data from water and rock analyses and from simple batch experiments conducted in the laboratory were used to evaluate the potential impact of geochemical factors on the scheme. Computerised geochemical modelling was used to anticipate the effects of mixing water types, potential mineral precipitation problems and the influence of the native groundwater in the breccia pipe on recovered water quality.

5.4.1 Effect of mixing water types

Mixing of different water types may drive several chemical reactions, leading to a final water composition that is different from a simple conservative mixture of the components in the two end-members. Redox and pH buffering effects are likely to be observed as the species in solution are redistributed among redox states and protonated or deprotonated to achieve a new equilibrium.

The computer program PHREEQC (Parkhurst 1995) was used to investigate the effects of mixing on the aqueous geochemistry of the injection system. Thermodynamic constants from the WATEQ4F database for natural waters were used for the calculations (Ball & Nordstrom 1991). In the mixing simulations, each of the recharge water sources was added to the breccia pipe water at mixing ratio increments of 10% and the equilibrium solution composition calculated, to investigate the range of solution compositions that could be found across a mixing front. Initially, only the effects in the aqueous solution were investigated as no solid phases were specified in the model. Redox conditions were approximated using estimated pe values of +3 for the breccia pipe and +2 for the recharge boreholes, based on average Eh readings for similar boreholes logged in the field. The default pe of +4 in PHREEQC was used for the surface water source. Simulations used field-measured values for pH and temperature. Predicted pH curves obtained from mixing Karoo groundwaters or surface water with the native breccia pipe water are shown in Figure 9.

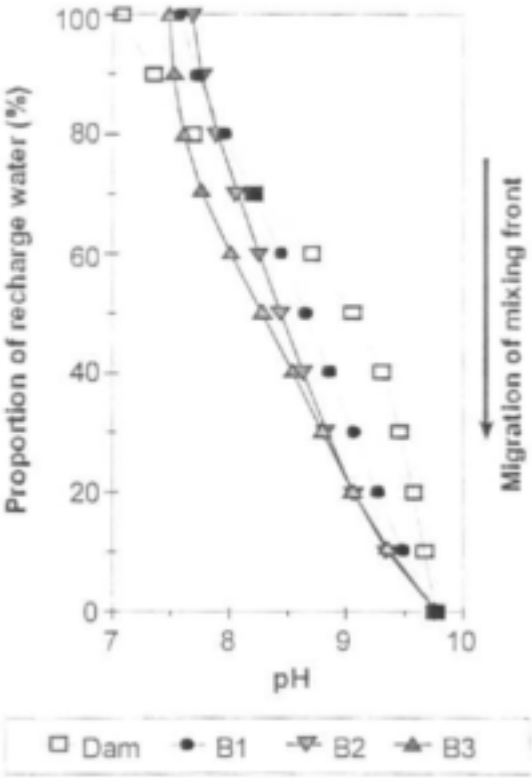

Figure 9 Simulated pH values for mixing of potential recharge sources with native breccia pipe water. Dam = Karee Dam + G39973; B1 = G39648 + G39973; B2 = G39638 + G39973 and B3 = G39627A + G39973.

Buffering by the carbonate system, in particular, but also by other proton-producing or consuming reactions, including redox reactions, will control the pH of the mixed waters. The prevalence of deprotonated species in the alkaline breccia pipe water will favour reactions that consume H^+ and shift the pH to higher values than those from a conservative mixing process. The effect is likely to be most pronounced for the dam water, which has the lowest alkalinity and lacks the buffering capacity imparted by the carbonate species in the other sources.

Results

- The mixing simulations showed that a high ratio of recharge water would need to be present in the recovered water to decrease pH values to acceptable levels (e.g. below pH 9 for Target Range for Domestic Water, DWAF, 1996). These high proportions of recharge water would defeat the objective of using the breccia pipe groundwater to decrease salinity by blending with the other municipal groundwater sources.

- Mixing ratios of 80 to 90% injected water (Dam or Karoo groundwater) are also needed to reduce fluoride concentrations to below the guideline level of 1 mg/L.

- Karoo groundwater sources were predicted to have a higher risk of mobilising As than recharge using surface water.

- Water from G39861 cannot be blended in the breccia pipe to produce acceptable levels of F^- and treatment would be required after recovery.

5.4.2 Potential mineral precipitation and the risk of clogging

Mineral saturation indices (SI) for a range of sparingly soluble solids were calculated during the mixing simulations to evaluate the potential for mineral precipitation. Saturation indices indicate the thermodynamic tendency for a mineral to dissolve (SI<0) or precipitate (SI>0), but do not account for the reaction rates of solid-solution phase changes.

Mixed solutions (Figure 10) show a greater degree of supersaturation with respect to calcite, ferrihydrite and fluorite than for the pure end-member solutions. Calcium carbonates (e.g. calcite) and iron oxide/hydroxides (e.g. ferrihydrite) pose a risk of filling pore spaces and thus reducing the available space for water storage. Fluorite precipitation, on the other hand, is a positive effect since it can decrease the concentrations of dissolved F^-.

The risk of calcite precipitation is greater if Karoo groundwater, rather than Karee Dam water is used for recharge. This was confirmed by a simple test tube demonstration where approximately equal proportions of water from G39973 and G39648, G39973 and G39638 or G39973 and G39627A produced a white precipitate from the original clear solutions. No precipitate formed from Karee Dam water and G39973.

Model sensitivity analysis showed that the calculation of carbonate mineral SI's was very sensitive to pH above pH 7. SI's for calcite, dolomite, aragonite and magnesite are increased significantly by raising the simulation pH from 8 to 10. Predicting the clogging potential of carbonate minerals, therefore, rests on the ability to anticipate pH changes in the AR scheme.

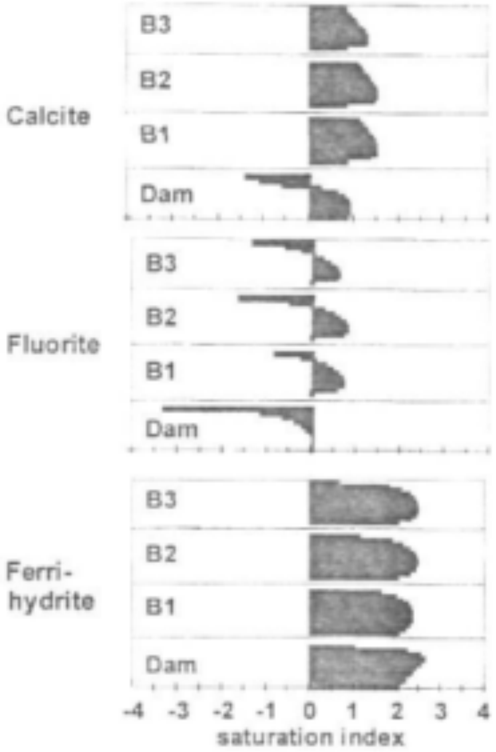

Figure 10 Saturation indices for calcite, fluorite and ferrihydrite calculated for water mixtures. Each simulation starts at 10% recharge water mixed with breccia water at the bottom of the graph and increases by 10% increments to 100% recharge water (all breccia water replaced) at the top. Each section of the bar graphs, therefore, represents the downward migration of a mixing front similar to that shown in Figure 9. Dam = Karee Dam + G39973; B1 = G39648 + G39973; B2 = G39638 + G39973 and B3 = G39627A + G39973.

Saturation indices of ferric hydroxide minerals decrease slightly with increasing pH in the pH 7 to 10 range. The solubility of minerals such as goethite and ferrihydrite is extremely sensitive to redox conditions. Saturation indices are below zero when negative pe values are used, because of the increased tendency for dissolved iron to occur as the more soluble Fe^{2+} form under reducing conditions. Without accurate knowledge of solution pe and *in situ* determination of both Fe^{2+} and Fe^{3+} concentrations, rather than total dissolved Fe, it is difficult to have any confidence that Fe oxides will precipitate, as is suggested by the saturation indices.

The potential severity of clogging by precipitation was assessed by rerunning the mixing simulations, this time allowing solid calcite, ferrihydrite and fluorite to precipitate. PHREEQC was used to calculate the mass of solid that would need to be precipitated per litre of water for the SI to reach zero (Figure 11).

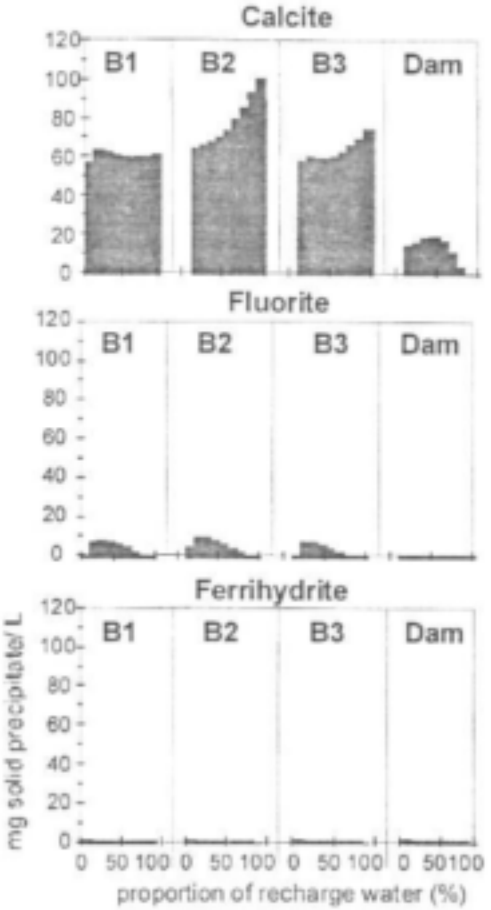

Figure 11 Calculated total mass of mineral solids that could precipitate from mixtures of breccia water with various recharge waters. Dam = Karee Dam + G39973; B1 = G39648 + G39973; B2 = G39638 + G39973 and B3 = G39627A + G39973.

Results

- Low total dissolved Fe concentrations mean that the mass of Fe(III) oxide that could precipitate is negligible, even though ferrihydrite saturation indices are higher than those of the carbonate minerals.

- Carbonate mineral clogging would appear to pose a far greater risk to the injection scheme than iron clogging in terms of the volume of material involved, especially when using the borehole recharge sources. Precipitation poses a risk for encrustation of pipelines and clogging of breccia fractures if the CO_2 partial pressure is too low to solubilise these minerals.

- Fluorite precipitation could remove up to 3.8 mg/L dissolved F^- by careful blending using Karoo groundwaters as recharge sources. Depending on the relative rates of precipitation, however, calcite, dolomite or aragonite precipitation could also remove calcium from solution, thereby inhibiting this natural removal of dissolved fluoride.

5.4.3 Water-rock interactions

The geochemical modelling calculations relied on the assumption that the subsurface reservoir is chemically inert and does not react with the mixing aqueous solutions. This is unlikely in the case of the breccia pipe, which contains reactive clay mineral surfaces and secondary mineral phases. Rock samples from the breccia pipe were crushed and reacted in laboratory batch-type experiments to investigate the potential impact of water-rock interactions on the quality of water that could be recovered from the breccia pipe after storage.

5.4.3.1 Leaching test

Bulk samples of three different particle size fractions (<63 mm, 63 mm - 2 mm and 2-10 mm) were leached with deionised water. A 333 g/L slurry of each sample was placed horizontally on a reciprocating shaker in a polyethylene container and shaken at 1 cycle per second for 500 hours to determine the products of mineral dissolution under atmospheric conditions (Table 5). An additional sample containing a concentrate of secondary hydrothermal minerals (2-10 mm fraction) was treated in the same manner (sample M1).

Table 5 Results of rock leaching experiments

Parameter	*Size fraction*			
mg/L	*<63 mm*	*63mm - 2 mm*	*2 - 10 mm*	*M1*
pH	8.5	8.6	8.3	7.4
Na	53	50	58	44
K	50	27	29	40
NH_4	7.5	2.4	3.6	5.7
Mg	7.0	7.4	3.2	18
Ca	16	19	8.9	46
F	0.8	1.0	1.2	5.4
Cl	9.4	5.1	6.4	4.5
NO_3	0.7	0.6	0.7	<0.1
SO_4	35	50	90	270
T.Alk (meq/L)	4.10	2.45	2.45	0.82

Results

- The rock samples contain significant amounts of soluble Na^+, K^+ and alkalinity and were also found to release NH_4^+, SO_4^{2-} and F^- into solution, all of which could be added to low salinity recharge waters.

- K^+, NH_4^+, F^- and SO_4^{2-} were readily leached from hydrothermal clay minerals, apophyllite and oxidized metal sulphides. The highest F^- concentration was leached from a sample rich in fluorapophyllite

- Sulphide mineral oxidation and addition of atmospheric CO_2 caused the pH to decrease with time.

- Alkalinity is generated by most weathering reactions (Stumm 1992) and increased with increasing solid surface area in these experiments.

Although the tests indicate potential ***trends*** in pH and dissolved species, the measured concentrations cannot be considered to reflect those expected in the field. The laboratory tests could not simulate several controlling factors that will be operative under field conditions. In reality, much longer reaction times, larger water to rock ratios, no artificial effects of rock crushing, different microbial populations and different pH, redox and gas compositions will be present in the breccia recharge tests.

5.4.3.2. Equilibration test

A 350-hour batch equilibration test was also undertaken, reacting the surface and groundwater recharge sources from Calvinia with a fine fraction (<63 mm) of bulk rock material from the breccia pipe, following the same procedure as the water-rock leaching tests.

Results

- Reaction of the low salinity dam water produced similar reactions to those found when leaching the rock with deionised water. A small, but significant quantity of As was also released to the dam water. The pH rose immediately to 9.3 on contact with the rock and then gradually fell to 8.0 over the course of the experiment.

- High solute concentrations in the groundwater sources dominated over any solutes added by the rock and buffered the pH at lower values than for the dam sample. Na^+, Ca^{2+} and Mg^{2+} concentrations decreased slightly in the brackish groundwater, by cation exchange (Na^+) and carbonate precipitation (Ca^{2+} and Mg^{2+}), but in all cases the removal efficiency was negligible and could not be considered a significant improvement in water quality.

- The proportion of NH_4^+ and K^+ in the groundwaters increased substantially after reaction with the finely divided rock.

- The artificial powdering of the rock material and limitations of working under laboratory conditions cannot be ignored when applying information from these batch tests to the borehole injection scheme.

5.5 Implications for the artificial recharge scheme at Calvinia

This section summarises the influence of water and rock chemistry, and the predicted fate of their mixing, on the pilot artificial recharge scheme. The three major concerns at Calvinia, namely loss of permeability, changes in water quality and potential health risks associated with using recovered water from the Kopoasfontein breccia pipe are discussed. Suggested treatment options, for the prevention of clogging and improvement of recovered water quality, conclude this section.

5.5.1 Potential loss of permeability

The Karoo groundwater sources from boreholes G39648, G39638, G39627A and G39861 are not recommended for borehole injection, because of their already inferior quality and the high risk of **carbonate precipitation** in the aquifer and scaling of pipelines and other infrastructure. Calcite cement already fills most of the fractures in the upper 40 m of the breccia pipe and injection with high Ca^{2+} and Mg^{2+} waters may precipitate further carbonate material unless the pH can be lowered or the CO_2 partial pressure raised to prevent this.

In comparison, the risk of clogging by **iron oxide precipitation** is low. Reducing conditions in the subsurface are not conducive to iron oxide precipitation. Low total dissolved iron concentrations also restrict the total mass of iron solids that could be precipitated from the groundwater. Fluorite precipitation is also not anticipated to be significant in terms of permeability loss.

Use of surface water sources for artificial recharge could lead to clogging of pore openings and fine fissures in the subsurface, if **suspended solids** are not completely removed from solution. For this reason, the use of treated water from the purification plant is planned for artificial recharge, rather than untreated Karee dam water, which contains suspended inorganic and organic particulates.

Swelling of clay minerals can reduce the effective porosity and permeability of an aquifer, by decreasing pore sizes and inhibiting the flow of groundwater. **Dispersion of clay particles** in an aquifer can also cause irreparable damage in an artificial recharge scheme if clay particles become detached and move into the aquifer pore-throat regions causing clogging (Pyne, 1995). Aquifer colloids (including organic matter as well as silicate clays) disperse most readily at low salt concentration, high exchangeable Na^+ levels and high pH (McBride, p.287). These conditions already exist in the Kopoasfontein groundwater and further destabilisation of aquifer colloids is unlikely. The swelling properties of clay minerals in the shale fragments of the Kopoasfontein breccia may reduce the primary porosity of the aquifer and lower the permeability of the shale matrix, thus restricting the flow of groundwater through the fine-grained shale and mudrock clasts. Secondary porosity, by virtue of the fractures and cavities in the breccia pipe is, however, by far greater than primary porosity and is not likely to be affected significantly by clay swelling.

Recharge with low salinity surface water might cause **dissolution of** some of the secondary **cementing material** in the breccia pipe, thereby dislodging the breccia fragments. It is, however, unlikely that the particles released in this way will cause a significant loss of permeability, since the shale fragments are probably too large to move with the groundwater and block pore openings. Recharge with Karoo groundwaters would be likely to cause carbonate mineral precipitation, rather than dissolution.

Evolution of gas bubbles by chemical reactions is also not expected to have a major effect on permeability, since gases should be dissolved or escape through the breccia pipe fractures. There is some indication that gases such as H_2S and CH_4 are already generated under reducing conditions in the aquifer. Dillon and Pavelic (1996) suggest that water recharged into anoxic aquifers, such as the Kopoasfontein breccia pipe, should be low in NO_3^- and organic C, to limit denitrification. In this respect, the Karoo groundwater sources are less likely to produce nitrogenous gases than the surface water. Care will need to be taken to minimise air entrainment and cascading in the injection boreholes.

5.5.2 Potential changes in water quality

Water recovered from the Kopoasfontein breccia pipe after artificial recharge is intended as a domestic water source for Calvinia. Chemical and biochemical processes that have an impact on the water quality can, therefore, play an important role in determining the success of the scheme. Changes in water quality include both the introduction of undesirable constituents to the recharged water from the native groundwater or host rock and improvement of water quality by immobilisation or dilution of contaminants in the aquifer or addition of beneficial elements.

5.5.2.1 pH changes

pH is a controlling variable for many of the potential reactions during recharge and should be measured regularly. Ionic speciation, mineral solubility, sorption, ion exchange and many biochemical processes are dependent on solution pH.

The geochemical system in the Kopoasfontein breccia pipe is naturally **alkaline**. Modelling and laboratory experiments indicate that the **pH** of artificially recharged water **is likely to increase**. The minerals in the breccia cause high pH conditions in waters with which they are brought into contact so that the natural groundwater has an *in situ* pH near 10 and low salinity waters achieve a similar pH between 9 and 10 when reacted with crushed rock samples. The dominance of Na and K over Ca and Mg in the system contributes to the evolution unusually high pH values. The typical pH range for systems buffered by equilibrium with Ca and Mg carbonate minerals is near pH 8. Reduction reactions under the anoxic conditions in the Kopoasfontein breccia pipe will also tend to consume H^+, causing the pH to rise.

The Dam and G39861 waters have low Ca and Mg and are more likely to evolve high pH conditions (above pH 9) than the groundwaters from G39648, G39638 and G39627A, which have better pH buffering capacity.

Over the **longer term**, the introduction of oxygenated surface water to the Kopoasfontein groundwater system may bring about a **gradual decrease in pH**. Two important processes of acidification were identified during batch test experiments, namely nitrification of ammonia and sulphide oxidation. Both these redox processes are sluggish, and unlikely to have an immediate impact on water quality, but they may be catalysed by biochemical processes once populations of microorganisms are established.

Effects of pH changes

Highly **alkaline conditions** have several negative implications for water quality. These conditions are generally favourable for the mobilisation of anions in solution, due to low anion sorption capacity and competition with OH^- for anion sorption sites. **Aluminium** solubility is enhanced by the formation of the anionic species $Al(OH)_4^-$ and higher concentrations of **F^-** and **As** oxyanions in the Kopoasfontein groundwater are likely to be present in recharged surface water under alkaline conditions. Any ammonium ions released to solution from the breccia rocks are also likely to be partially deprotonated to the more toxic **NH_3** form in high pH waters.

Decreasing pH with time after recharge is likely to be associated with increasing concentrations of SO_4^{2-} and possibly NO_2^- or NO_3^- in the groundwater. The rocks contain sufficient alkalinity, however, to neutralise any acid generated by sulphide oxidation or nitrification and the high pH should **immobilise heavy metals** that might be released.

5.5.2.2 Redox changes

Oxidation-reduction reactions control the chemistry of multivalent elements such as Fe, As, S, Cu, Mn, Se and N in the system. Many of these elements have a significant impact on the quality of water for domestic use.

There is evidence that a **reducing environment** exists **in the breccia pipe**, based on the presence of NH_4^+ ions, H_2S and CH_4 gases and low or undetectable dissolved O_2 concentrations in the groundwater. Groundwaters from G39861 and G39648 also contain H_2S and low dissolved O_2. Recharge with these sources is unlikely to cause a significant change in redox conditions, particularly since the redox levels will potentially be buffered by the presence of reduced sulphide minerals in the Kopoasfontein rocks.

The Karee **Dam water**, however, **is oxidising**, containing considerable dissolved O_2, and is likely to have significant redox buffering. Mixing of waters from completely different redox environments may cause several changes in the speciation of redox elements in the Kopoasfontein pipe as the system moves towards a new redox equilibrium. The kinetics of redox reactions, however, are often slow, so disequilibrium is common.

Effects of oxidation

Oxygen injected into the breccia pipe along with recharged surface water may initially be consumed by bacterially mediated oxidation of organic matter, sulphides or ferrous minerals. Ammonium in the Kopoasfontein rocks and groundwater might be expected to undergo nitrification, **producing NO_2^- or NO_3^-** in solution if conditions are suitable for the growth of the microorganisms that catalyse these reactions. Oxidation of sulphide minerals will **increase** the concentrations of **SO_4^{2-}** and decrease the pH if conditions become sufficiently oxidising. Sulphur in its S(-II) form is unstable when the solution redox potential is buffered by nitrification reactions.

Oxidation may improve the quality of the recovered water, by **lowering the concentrations of arsenic** and dissolved iron below those that can be achieved by dilution alone. Conversion of As(III) to As(V) oxyanions and a decrease in pH is likely to increase the chemisorption of arsenic species to **iron oxide minerals** which **precipitate** in alkaline, oxidising environments.

5.5.2.3 Changes in salinity

Recharge of surface water from the Karee Dam with low salinity and dominant Ca^{2+}-Mg^{2+} cation signature is likely to result in a **slightly higher salinity**, Na^+-dominated water when recovered from the pipe.

The brackish Karoo groundwaters have a greater concentration of ions in solution than the native breccia pipe groundwater. Their solute content is already high and is not increased significantly by reaction with the rocks. There is even some suggestion that Na^+, Ca^{2+} and Mg^{2+} concentrations in these solutions are decreased on reaction with the rock by mineral precipitation, adsorption or ion exchange processes. The effect is negligible, however, and does not constitute a noticeable improvement in the quality of these waters. It is **unlikely that a potable water supply can be achieved by blending the Karoo waters with the *in situ* breccia water**. No suitable mixing ratio could be found that would allow sufficient decrease of both the salinity from the Karoo waters and the fluoride in the breccia water.

5.5.2.4 Changes in dissolved species

The breccia rocks have the potential to release significant amounts of NH_4^+, SO_4^{2-} and F^- into solution over a long time period. It is suspected that these species are mainly derived from the hydrothermal minerals in the breccia.

Arsenic and fluorine concentrations in rock samples are equivalent to, or lower than, typical values for shales, but the alkaline, reducing conditions in the Kopoasfontein groundwater are favourable for solubilising these components. Reacting the Karee Dam water with the Kopoasfontein rocks produces similar high pH conditions to those in the breccia pipe. **High As and F^-** might, therefore, be expected to jeopardise the quality of artificially recharged surface water recovered from the breccia pipe, **if the storage time is long** enough and if the organic matter content high enough for sufficiently reducing conditions to develop. When reacted with the Karoo groundwaters however, the pH-buffering effects of Ca^{2+} and Mg^{2+} mineral precipitation may prevent F^- and As from reaching problematic concentrations in solution. Precipitation of CaF_2 may also help to lower F^- concentrations.

Laboratory investigations of the dissolution of fluorapophyllite were undertaken to assess the risk of fluoride release from the breccia minerals. These experiments showed that fluoride is released at a very slow rate of 1×10^{-11} mol/m^2/s (Cavé *et al.*, 2001). Assuming a total surface area between 1000 and 10 000 m^2 for all the fluorapophyllite in the breccia, 20 000 m^3 of water could safely be stored for anything from 350 to 3500 years before the F released from the mineral exceeded the recommended limit of 1 mg/L. This is a conservative estimate, since laboratory-measured rates are typically much faster (up to several orders of magnitude) than field-measured dissolution rates. Fluoride and arsenic released from the minerals are likely to be insignificant in comparison with that already present in the natural groundwater because of the slow rates of silicate and sulphide mineral dissolution.

Mixing reactions between injected and native waters are expected to **dominate over water-rock interactions**, because of the low water-rock contact area and short time scales involved.

5.5.3 Health implications of using recovered water from the scheme

Several constituents of the possible recharge source waters and native groundwater were identified which could be of concern in terms of their potential health risk for domestic users. These include F^-, As, Se and H_2S. **Fluoride and arsenic** possibly pose the **greatest risk**, in that these are naturally derived from the Kopoasfontein rocks and present in the native groundwater at concentrations high enough to cause serious health problems for long-term users.

Fluoride concentrations over 8 mg/L are high enough to cause not only irreversible dental fluorosis, but also skeletal fluorosis in individuals who consume the water for an extended period (Boyle and Chagnon, 1995). Chronic As toxicity can cause pigmentation disorders, keratosis or peripheral vascular disorders, but there is also evidence that arsenic is carcinogenic and may cause skin cancer as well as a number of internal cancers (Edmonds and Smedley, 1996). Human health is seriously at risk if arsenic concentrations exceed 0.2 mg/L.

The South African Water Quality Guidelines (DWAF, 1996) recommend **target ranges of 0 to 1.0 mg/L fluoride and 0 to10 mg/L arsenic** for domestic supplies. Care would need to be taken that sufficient quantities of low salinity surface water are injected to dilute the natural

F and As in the groundwater to below these levels.

Fluoride poses a lower risk of reaching undesirable concentrations if the Karoo groundwater sources are used for artificial recharge, since these contain high Ca^{2+} which is known to control fluoride solubility (e.g. Handa, 1975; Boyle, 1992). The greater pH buffering capacity of the Karoo groundwaters should also keep the pH at lower levels than for recharged surface water and prevent mobilization of additional F or As.

Selenium concentrations measured for G398638 (26 µg/L), G39648 (23 µg/L) and G39627A (55 µg/L) were slightly greater than the 20 mg/L level recommended for domestic water (DWAF, 1996). These would be diluted by mixing during artificial recharge and Se(VI) may also be converted to the less mobile Se(IV) form in the reducing environment of the breccia pipe. **Selenium** is considered to pose a **low risk** for users of the recovered waters.

The Karoo groundwaters, however, also contain an **objectionable quantity of dissolved solids** that will not only impair the taste of the water, but also potentially cause short-term adverse health effects by disturbance of the salt balance, unless diluted several times with low salinity water.

The breccia pipe rocks are unusually rich in nitrogen and rock leaching tests and water-rock equilibration experiments produced NH_4^+ in solution. Although this ion is not toxic, the dissociation of NH_4^+ to NH_3 at high pH and biological transformations of NH_4^+ could lead to problems for domestic water supplies in high enough concentrations. Ammonia is particularly toxic to aquatic organisms (Dallas and Day, 1993) and if conditions in the aerobic zone of the pipe, or after recovery become sufficiently oxidising, nitrification reactions may convert NH_4^+ to the oxyanions NO_2^- or NO_3^-. Nitrite and nitrate pose a health risk to bottle-fed infants at high concentrations. The guideline limits set by the WHO are 10 mg/L as N (45 mg/L NO_3^-) for nitrate and 1 mg/L as N (3.3 mg/L NO_2^-) for nitrite (WHO, 1984). The clay minerals in the shales, however, provide a good natural mechanism for the immobilization of NH_4^+ by sorption or ion exchange and **N species** expected to pose a **low risk** in terms of health problems related to artificial recharge.

5.5.4 Possible treatment options

Subjecting the recovered water to the traditional treatment practices of flocculation, filtration and chlorination at the municipal water purification plant may alleviate some of the potential quality problems, but there are certain constituents that require special consideration. Some alternatives available for prevention or treatment of clogging and water quality problems are summarised briefly in Table 6.

In general, pH neutralization and the introduction of oxidizing waters to the breccia pipe may prevent some of these problems from reaching the stage where treatment is required. To lower the pH in the breccia pipe, the injection water could be saturated with CO_2 to increase its acidity and decrease its pH before recharge. This will be more effective if the dam water is cold, since greater concentrations of CO_2 will dissolve. Injection into the warmer breccia pipe may, however, lead to some degassing of CO_2 and the problems associated with gas generation in the subsurface. Additional pH neutralisation by dosing with mineral acids, acid coagulants (e.g. $Al_3(SO_4)_3$ or $Fe_3(SO_4)_2$) or CO_2 gas is still likely to be required once the stored water is recovered.

Table 6 Possible preventative or treatment options for water quality and clogging problems that may arise from artificial recharge at Calvinia

Potential problem	*Prevention/treatment option*	*Comment*
Water quality problems		
High pH	Addition of acid (H_2SO_4, HCl, CO_2 gas).	Metal coagulants e.g. Al^{3+} or Fe^{3+} will also lower the pH.
High fluoride	Pretreatment with Ca (e.g. lime, gypsum) or Al^{3+} salts followed by ion exchange using activated alumina or reverse osmosis.	Pretreatment will only reduce high F (>4 mg/L), but will not treat F to domestic guidelines.
High arsenic	Oxidation to As(V) by Cl_2 or $KMnO_4$ followed by conventional coagulation (Al^{3+}, Fe^{3+} salts or lime) flocculation, settlement & filtration.	Most effectively removed in pentavalent form.
High EC	Desalination e.g. electrodialysis, reverse osmosis, distillation or ion exchange.	Expensive desalination can be avoided if sufficient low salinity water can be blended.
Formation of ammonia	pH neutralisation.	
Formation of nitrate	Addition of Cl salts or bactericide. Treatment by anion exchange or biological reduction using a carbon source e.g. ethanol.	Biological approaches involve inhibition of nitrification or promotion of denitrification.
Clogging problems		
Carbonate solids	pH neutralisation. Avoid recharging large volumes of very high Ca- or Mg-rich waters.	
Ferric iron solids	pH adjustment.	Not likely to occur in high volumes.
Biofouling	Use low DOC water and disinfect (e.g. Cl_2) before injection.	
Injected solids	Adequate filtration before injection. Mechanical redevelopment.	
Clay dispersion	Addition of flocculant e.g. $CaCl_2$.	Divalent cations help to alleviate clay swelling.
General clogging	Mechanical redevelopment by pumping, compressed air or water jetting or backwashing.	

5.6 Geochemical conclusions and recommendations

The release of F^-, As and possibly NH_4^+ from the rocks in the Kopoasfontein breccia pipe is likely to cause problems for the managers of the artificial recharge scheme, the severity of which will depend on the duration of storage and geochemical conditions in the subsurface.

The treated dam water is the most suitable recharge source if high mixing proportions can be maintained by dewatering most of the natural groundwater from the pipe before recharge. Geochemical conditions need to be monitored regularly after injection. Recommended parameters for monitoring to protect the stored water quality include pH, EC, dissolved oxygen, F^-, SO_4^{2-}, K^+, NH_4^+ and As concentrations. Monitoring of these parameters during pilot injection experiments may help to identify problems at an early stage so that they can be addressed before valuable water resources are lost.

The injection of waters which decrease the pH of the system towards neutral and introduce oxidising agents is encouraged as a means of improving the water quality. Surface water from the dam contains dissolved oxygen which will enhance both pH neutralisation and oxidation, although the pH change is likely to be small. Oxidation and acidification will shift chemical equilibria that may help to control F^- and As concentrations and decrease $CaCO_3$ precipitation in the pipe. The same geochemical trends, however, could also favour nitrification of NH_4^+ to NO_2^- or NO_3^- and the generation of SO_4^{2-} from sulphide mineral oxidation.

The water recovered from the breccia pipe should be subjected to the same water purification procedure as the water usually supplied by the town dam i.e. addition of $Al_3(SO_4)_3$, liming, filtration and chlorination. Dosage of the various chemicals may need to be adjusted to suit the conditions of the recovered water.

Note:

These conclusions are based on geochemical modeling and laboratory experiments. The usefulness of any modelling exercise is limited by uncertainties in the input data and the modeller's conceptual understanding of the hydrochemical system. Laboratory experiments are also restricted by the inability to artificially reproduce all possible controlling variables of the natural environment. These inexpensive methods allow preliminary assessments to be made of the feasibility of recharge activities, but cannot replace the value of pilot testing in the field.

6. THE FIRST INJECTION RUN

6.1 Injection volume and water level rise

Injection started on 10 April 2001, after the pipe had been dewatered to 142.6 mbgl, and the injection was completed on 4 July 2001. Approximately 31 800 m^3 was injected at an average rate of about 4.6 L/s. This figure is not precise because the flow meter on one of the injection boreholes was initially faulty. The injection was stopped once the water level reached 26.5 mbgl. Figure 6 shows the water rise with injection.

The maximum injection rate achieved was 6.9 L/s during an uninterrupted 10 day period. If the pipeline can take a greater flow rate, this could be increased to about 10 L/s, which is the surplus available water from the water treatment works during winter. The capacity of the water treatment works is 1 920 m^3/day (22 L/s), and during winter, it will be able to supply 800 m^3/day for injection, since the balance is needed for the town's water supplies.

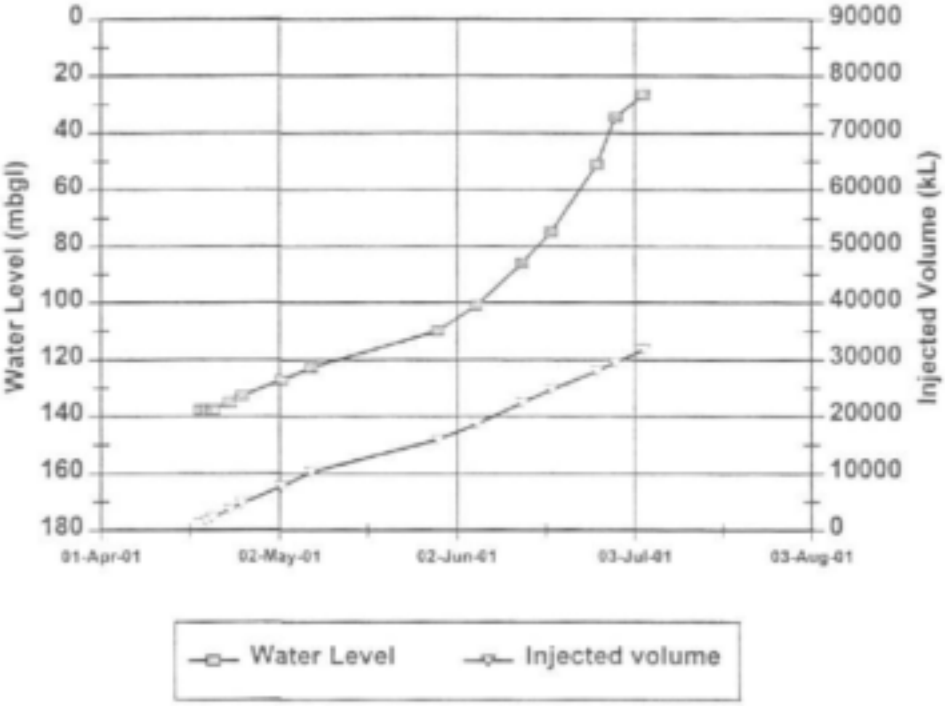

Figure 12 Water level rise with injection

The injected volume was 16 000 m^3 less than the abstracted volume. This is partly because during the initial stages of injection the flow was not accurately measured (due to a faulty meter); partly because the injection was stopped about 8 m above the natural rest water level; and partly because water did leak into the pipe from the dolerite above during the long time it took to bring the water level down to the pump.

6.2 Water quality changes after injection

Vertical changes in electrical conductivity and pH of the groundwater in the breccia pipe were logged with the aid of a HydroLab® MiniSonde® 4a water quality multiprobe and datalogger. The multiprobe has pressure sensors to measure depth below the water table and electrodes for the measurement of pH and electrical conductivity. A Freshflow™ circulator is used to stir the water for improved reproducibilty of readings. The instrument is capable of collecting data at depths from 0 to 200 m below the water table. The multiprobe was lowered down accessible boreholes at the site both during the dewatering of the pipe and after the first round of surface water injection. The results of the logging are shown in Figure 13 and 14.

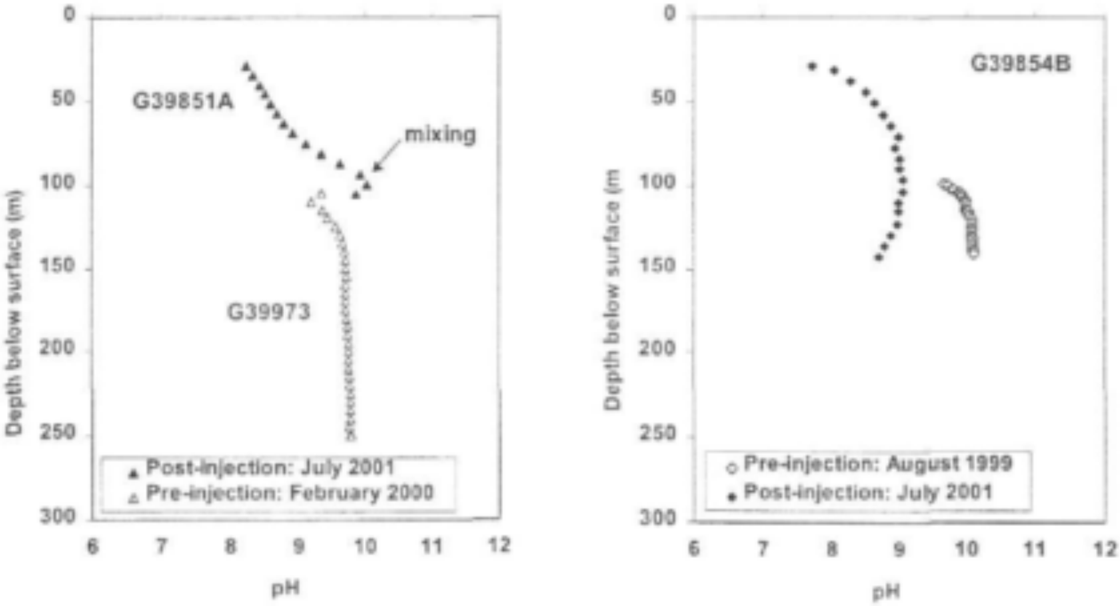

Figure 13 Variation in pH with depth a) down boreholes G39973 and G39851A in the centre of the breccia pipe and b) down borehole G39854B towards the edge of the pipe, near to the injection borehole.

Injection of surface water has decreased the groundwater pH from its natural value of approximately pH 10 to near pH 8 in the upper portion of the pipe. The initial surface water pH is near 7, but mixing with the alkaline breccia water and strong pH buffering reactions, cause a steep increase in pH with depth, particularly below 70 m from the surface.

The shift to lower pH represents an improvement in the groundwater quality, since problems such as high fluoride concentrations are partly attributable to the high groundwater pH. Long term monitoring of pH in the breccia pipe is recommended. The pH may continue to rise slowly with time as chemical equilibria disturbed by the injection are re-established.

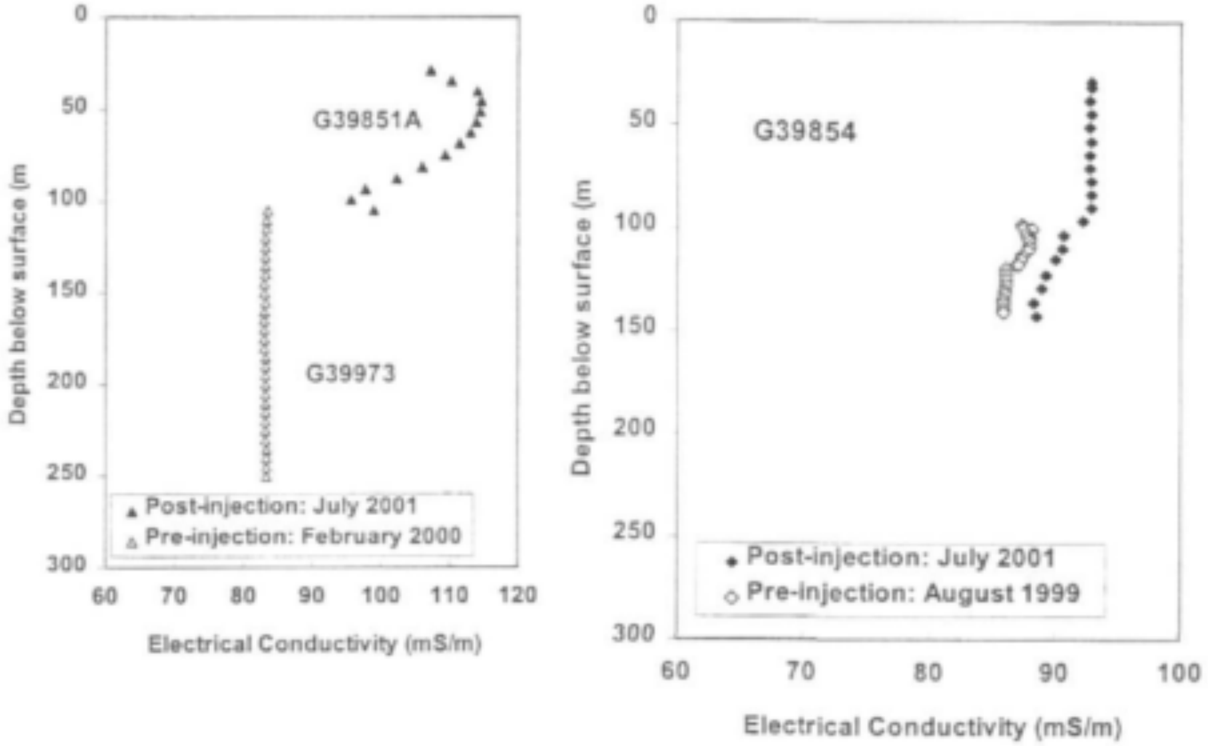

Figure 14 Variation in electrical conductivity (EC) with depth a) down boreholes G39973 and G39851A in the centre of the breccia pipe and b) down borehole G39854B towards the edge of the pipe.

The electrical conductivity logs show that the salinity of the groundwater has increased after the injection exercise. The natural groundwater EC is between 80 and 90 mS/m, but has increased to over 100 mS/m in the upper, central part of the pipe. The increase in salinity appears to be less pronounced towards the edges of the pipe.

The surface water is a low salinity water (EC ~ 20 mS/m) and so the additional salinity is not brought in by injection. Instead, the injection activities may be dissolving mineral salts which have precipitated in the fractures of the breccia near to the surface. Precipitated calcite is particularly common in the upper 40 metres of the pipe. Natural salinity, caused by evaporative concentration, is common in the soils of semi-arid and arid climate zones. Salts may also precipitate in the capilary fringe above a fluctuating water table. These potential sources of salt could have been dissolved as the water table rose during filling of the pipe, leading to the increased groundwater EC post-injection. A better understanding of the salinity source may be gained by analysing the major ion chemistry of the recovered water to establish which ions are responsible for the effect.

The first injection run showed that it may take a few injection-recovery cycles before pipe has been sufficiently flushed to store fresh water for lengthy periods. At the time of writing this report, the second abstraction-injection run was being implemented.

7. CONCLUSIONS AND RECOMMENDATIONS

About 12 km east of Calvinia is a roughly cylindrical brecciated pipe that contains open fractures and cavities which could be used for sub-surface water storage. The Kopoasfontein breccia pipe is intensely fractured and brecciated to a depth of 250 m. Its full storage potential is not known because to date it has only been dewatered to 142 mbgl. Its usable storage depends on the depth of the column of the pipe that is utilised, and geophysical evidence suggests that porosity is highest up to 182 mbgl. From abstraction and geophysical data, the volume of water available up to a depth of 182 m is estimated to be 80 000 m^3, and up to a depth of 255 m, after which the porosity becomes very low, it is estimated to be 105 000 m^3.

The town's maximum winter consumption is 1 100 m^3/day, and their maximum summer consumption is 1 600 m^3/day. The reserves in the breccia pipe are thus equivalent to at least two to three months supply.

The existing pump can unfortunately only draw the water level down to 142 m. The available water up to this level is 48 000 m^3. If the municipality had a pump that could draw the water level down another 40 m, they could add another month's back-up, or more, to their water bank.

The ambient water quality in this geologic structure is unsuitable for human consumption, primarily because of the high fluoride and arsenic concentrations. If this water were to be used for domestic consumption over short periods, it would be necessary to dilute it with dam water at a ratio of at least 1:8 (Appendix 2.2).

Treated dam water is the most suitable recharge source. The residence time of injected dam water is affected by the unusual geochemistry of the breccia pipe system. At the time of writing this report the first injection run had just been completed. Geochemical conditions will need to be monitored on a regular basis from now onwards until the pattern of water quality changes is established. Recommended parameters for monitoring include pH, EC, dissolved oxygen, F^-, SO_4^{2-}, K^+, NH_4^+ and As concentrations.

The residence time of the stored water will depend on the process of blending with deeper groundwater within the pipe and on water-rock interactions. With repeated injection runs, it is expected that the residence time of stored water in the pipe will increase. Whether or not it will eventually be possible to use this water without any blending with dam water prior to consumption, is at this stage unknown.

It is, in any case, recommended that the stored pipe water be pumped to the dam prior to consumption, and not the mixing chamber before the reservoir. The reason for this is that the concentrations of the elements of concern may decrease due to the oxidising conditions in the dam and/or the modified chemical equilibrium after dilution with the dam water. If this is not possible, the stored pipe water should be allowed to run through the full treatment process (at the water treatment works) prior to consumption.

Injection conditions which decrease the pH of the system towards neutral conditions and introduce oxidising agents should be encouraged to improve the water quality. These will shift chemical equilibria that may help to control F^- and As concentrations and decrease $CaCO_3$ precipitation. The dam water contains oxygen and will assist in lowering the pH (slightly) and increasing the redox potential of the system. The pilot recharge tests should determine the amount of additional pH adjustment required. This can be achieved by dosing with acids or CO_2 gas, either before injection or after recovery. The same geochemical trends of increasing acidity and oxidation , however, could also favour nitrification of NH_4^+ to NO_2^- or NO_3^- and the generation of SO_4^{2-} from sulphide mineral oxidation.

In order to take advantage of this sub-surface reservoir, it is recommended that a pump capable of pumping a 180 m head be obtained, and that the breccia pipe be artificially recharged after each time it has been drained. The water used for artificial recharge should be fully treated dam water with very low turbidity.

With appropriate management and the right size pump, Calvinia could have approximately 80 000 m^3 of water reserves stored in a sub-surface reservoir for emergency needs.

It is imperative that the water abstracted from the pipe for consumptive use be sampled on a regular basis. The following analyses are recommended:

- pH and EC, daily;
- F^-, SO_4^{2-}, NH_4^+ and As, every two weeks.

The determinands Al, B, Fe, Pb and Hg should be tested at the start of each recovery period and only be included in the two week sampling runs if the initial values fail DWAF guidelines. A geochemist must assess these results, and advise on whether the water is suitable for consumption, or whether it needs to be diluted with dam water (and at what ratio). The stored water should preferably be pumped back into the dam, or alternatively, to the inlet of the water treatment works, prior to consumption.

Recommendations by Dr. Peter Dillon and Project Team Actions/Comments

Several helpful recommendations were made by Dr. Peter Dillon to improve the understanding of the geochemical processes and water quality issues, as these present the major challenge for the Calvinia scheme. Unfortunately, many of these suggestions involved detailed laboratory and analytical work which were not included in the original project outline and could not be incorporated within the time and budget constraints of the project.

Of the recommended actions, the conduction of field testing at the site was chosen as a priority over further laboratory studies. It was felt that the actual recharge trials would provide more valuable information in getting to grips with the issues of the scheme than trying to anticipate these from additional column tests or detailed mineralogical analysis. The first injection run was completed during the final months of the project and an agreement is in place between the CSIR, Hantam Municipality and the WRC to continue field trials for a futher year before operational guidelines can be drawn up for the management of the scheme.

The following comments are made in response to individual recommendations made by Dr. Dillon:

"*The field injection and storage experiment will require water quality monitoring that allows some interpretation of the processes and it is strongly recommended that organic carbon, dissolved inorganic carbon, nitrogen, phosphorus, sulfur-34, C-14, and C-13 be included in the monitoring program. Oxidation of injected organic carbon can increase the partial pressure of CO_2, and hence changes acid-base and redox reactions within the aquifer, and consequently shifts the equilibrium products.*"

The samples recovered from the first injection run have been analysed for pH, EC, F, Ca, SO_4 and Cl. Analyses of As, P, NH_4, NO_3 and DOC will also be conducted on selected samples from this test and a planned second injection run. Down-hole logging of pH and EC is also planned for the field testing phase.

It is felt that it would be difficult to justify the expense of carbon and sulphur isotope analyses at this stage, particularly since several analyses would be needed to define the end members of the system. Mineral samples would have to be analysed in addition to water samples to determine these constraints on the geochemistry. Cavé (in prep) found that carbon isotopes gave inconclusive information on the groundwater ages in the area, because of uncertainties in the input values and complex carbonate chemistry. The contribution and isotope composition of calcium carbonate and soil zone CO_2 are particularly difficult to establish in this case. Facilities for these analyses are also rare in South Africa and long turnaround times for analyses may be expected.

This would, nevertheless, provide an excellent research opportunity for a student in isotope geoscience who could undertake their own analyses, if one could be found.

"In preparation for the field injection, it is recommended that two laboratory column experiments be performed with subsamples of the same crushed (but not powdered) rock. (Mineralogy of the remaining subsample would be evaluated.) These column studies would be undertaken under anaerobic and intermittently aerobic conditions to represent conditions in the permanently and intermittently saturated zones respectively. It is suggested that continuously recirculating water be cycled through each column (with an anaerobic reservoir to allow water sampling and EC and DO probes) until water quality parameters become stable. Each column should be initiated by running breccia groundwater through until water quality is stable. One column should then be allowed to drain and be exposed to air (for a minimum of 3 weeks). Then surface water would be used to displace air and groundwater and recirculated through both columns until equilibrium, and water quality parameters measured, including isotopes. The aerobic column would then be drained and left exposed to air, while the other column would be purged with native breccia groundwater, until the experiment was repeated by recommencing the circulation of surface water through each column. The anaerobic column would need to be maintained under anaerobic conditions at all times (representing conditions in the aquifer beyond the immediate vicinity of the injection well). It is expected that each of the column phases should be at least 3 weeks to allow for microbial populations to grow under each new set of environmental conditions, as micro-organisms may mediate reactions that influence water quality, eg SO_4 reduction. The experiment can be extended if successive injection and storage cycles give results that indicate trends. At the end of the experiment the mineralogy of column materials should be reanalysed, and a mass balance performed for at least As and F.

Such experiments do not answer the question on the degree of exposure of minerals participating in reactions and the relationship between this in the columns and that in the field. However this type of experiment is a step closer to providing forecasts for the field experiment, and may assist with interpretation of the field experiment, whether the results are similar or contrasting. They may also suggest solutions such as, use of packers to recover water from within a depth range that excludes recovery of deep predominantly native breccia groundwater and shallow water that may have been subject to oxidation and is possibly enriched in As and F."

Such experiments were not performed due to lack of time and resources. As stated above, these experiments would not answer the questions regarding the mineral-water-air interactions fully. Consequently, these were considered of lower priority than conducting the actual field trials within the limited time and budget.

"A mass balance for As and F could possibly be estimated now and based on breccia mineralogy. This, together with the column experiments could assist in estimating the flushing volumes that will be required apriori. Even if these look large, it is suggested that the field experiment should proceed, as there are many uncertainties associated with the accessibility of the As and F for leaching."

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The source of arsenic is expected to be in the metal sulphides, but this has not even been established conclusively. Fluoride in the apophyllite has been investigated in great detail, but thermodynamic and kinetic constraints indicate that there may be other sources of F present (Cavé, in prep.), so even if the apophyllite could be quantified it may not give the required information for mass balance calculations.

If mass balance calculations were possible, their value would still be doubtful, since it is the mineral reactivity, rather than the bulk composition which will control water quality. The possibility of flushing all the F and As from the system is more a question of flushing the resident groundwater, which already contains these elements, than of flushing all F and As minerals out of the rock. Once the poor quality native water is removed, it is the relatively short residence times that are hoped to prevent the stored water from dissolving equivalent amounts of F and As from the breccia.

"It is recommended that consideration be given to water treatment alternatives for F and As removal, and if this requires experimentation, to establish a pilot plant if possible prior to recovery of the water to be stored in the breccia. This would enable this to have a beneficial use, in the event that it exceeded guideline values, and would facilitate further water being made available to the trial in the event that the first water recovered is not of suitable quality."

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APPENDIX 2.1:

Geological logs

BOREHOLE: **G39973**
DRILLER: RC Drilling (Smith's Mining) J van der Merwe
COORDINATES: Latitude 31.48536, Longitude 19.90031 (GPS 2D, Accuracy = 100m)
DRILLING DATES: 08-03-1994 to 30-03-1994
TOTAL DEPTH: 360 m
COLLAR ELEVATION: 1065.316 mamsl
WATER LEVEL (16-05-1994): 16.10 m

Depth (m)	**Lithology**
0-8	Baked, metasomatised grey/green mudstone and siltstone. *Brecciated - vugs filled with calcite.* Weathered/jointed to 6 m, thereafter weathered on joint surfaces only.
8-23.5	Melanocratic, very fine to fine grained dolerite. Weakly jointed and weathered to 10.5 m. Weakly jointed 20-21 m.
23.5-24	Metasomatised/brecciated, highly baked, light grey siltstone.
24-31	Highly baked, metasomatised, brecciated, grey/maroon mudstone. *Breccia vugs filled with calcite + chlorite.* 26-29 m mixed mudstone and melanocratic, very fine grained dolerite.
31-49	Baked, metasomatised, brecciated, dark grey to black mudstone and shale. *Minor open breccia vugs filled with calcite and chlorite.* Minor apophyllite and pyrrhotite. 35-39 m - weakly jointed (blocky).
49-51	Highly baked, metasomatised, brecciated, grey/maroon mudstone (hornfels).
51-60	Baked, metasomatised, dark grey to black shale. *Minor open breccia vugs filled with calcite + chlorite.* 60 m - jointed (blocky).
60-61	Baked, metasomatised, grey/maroon mudstone.
61-81	Baked, metasomatised, dark grey to black mudstone and shale. *Minor breccia vugs filled with calcite + chlorite.* 66-67 m weakly jointed (blocky). Minor thin zones of grey/maroon siltstone. 74 m - brecciated/jointed (blocky). Minor pyrrhotite.
81-82.5	Baked, metasomatised, highly brecciated, light grey to maroon/grey siltstone. *Breccia vugs filled with calcite +chlorite.*
82.5-87	Baked, metasomatised, dark grey to black shale.
87-217	Highly baked, metasomatised, brecciated, mottled, maroon/grey mudstone to light grey-grey/green siltstone. *Open breccia vugs, lined with chlorite and apophyllite, minor pyrrhotite.* 96-165 m highly brecciated, *abundant open vugs- abundant chlorite, apophyllite and pyrrhotite.* 175 m onwards, only *minor open breccia vugs*, generally hairline erratic fracturing and shattering, filled with chlorite. 196-197 m - *open vugs.*
217-242	Baked, metasomatised, remobilised, dark grey/maroon to grey/green, mottled siltstone and very fine grained sandstone. *Minor open breccia vugs*, lined with chlorite. Minor dark grey mudstone. 232-236 m highly baked creamy white shale.
242-305	Highly baked, metasomatised, brecciated, mottled, grey/green to maroon/grey mudstone. *Very minor open breccia vugs.* 279 m - well brecciated. Hornfels texture to 279 m
305-360	Baked, metasomatised, dark grey to grey/green shale.

Data source: Woodford, pers. comm.

BOREHOLE: **G39852**
DRILLER: Department of Water Affairs and Forestry
COORDINATES: Latitude 31.4851, Longitude 19.9012
DRILLING DATES: 06-06-1992 to 10-06-1992
TOTAL DEPTH: 150 m
COLLAR ELEVATION:
WATER LEVEL (11-06-1992): 27.700 m (0.1m casing elevation)
(09-07-1992): 26.030 m

Depth (m)	Lithology
0-0.5	Red/brown clay silt to very fine sand.
0.5-8	Weakly weathered/jointed, grey/green mudstone. Thin (<0.2 m) highly weathered horizons, clay in joints.
8-15	Slightly weathered/jointed, dark grey/black shale. Weakly baked.
15-21	Baked, weakly jointed, dark grey to grey siltstone. Maroon discolouration. *Pyrite* staining.
21-32.5	Melanocratic, very fine to fine, ophitic (speckles feldspar) dolerite. Slightly jointed.
32.5-33.5	Baked, light grey/green siltstone. (Breccia pipe?)
33.5-36	Mesocratic, fine-grained dolerite. Appears metasomatised.
36-55	Weakly baked, dark grey to black shale. Minor thin (<0.5 m) silty layers.
55-62	Weakly baked, dark grey mudstone.
62-150	*Breccia pipe* material. Light grey/green to grey/green/maroon siltstone. Well baked. 63 m - well jointed (blocky, ⌀ 60 mm). 66 m - jointed (blocky, ⌀ 20 mm), minor *calcite* in joints. 68 m onwards - metasomatised. 71 m - jointed (blocky, ⌀ 20 mm). Individual fractures, rather than fracture zones. 73-74 m - minor *calcite* + *pyrrhotite*. 74 m onward - hornfelsic texture. 75 m - well jointed (blocky, ⌀ 20 mm). 80 - 86 m - mineralised, *pyrrhotite* (max ⌀ 3 mm). 88 m - jointed, *apophyllite* + *pyrrhotite*, thin (10 mm) interbedded very fine quartzite. 96 m - hornfels, *apatite (?crystal)* with alteration halo, *calcrete-like* matrix. 99-122 m - hornfelsic, grey/maroon siltstone. 125 m - weakly jointed + *apophyllite*. 133 m - as per 96 m, weakly jointed. 136 + 138 m - weakly jointed (conchoidal) + minor *apophyllite* on joint surfaces.

Data source: Woodford, pers. comm.

BOREHOLE:	**G39854**
DRILLER:	Department of Water Affairs and Forestry
COORDINATES:	Latitude 31°29'08.3", Longitude 19°54'05.1"
DRILLING DATES:	08-07-1992 to 20-07-1992
TOTAL DEPTH:	150 m
COLLAR ELEVATION:	1064.353
WATER LEVEL (24-07-1992):	20.785 m
(21-08-1992):	26.555 m

Depth (m)	Lithology
0-1	Weathered/jointed, mesocratic, fine-grained dolerite.
1 -5.8	Baked, weathered/jointed olive green mudstone.
5.8-19	Melanocratic, very fine to fine grained dolerite.
19-20	Baked, light grey/maroon siltstone.
20-25	Weakly baked, dark grey mudstone.
25-150	Dark grey to black shale. Slightly baked to 29 m. Minor thin (<0.5 m) silty horizons. 67 m onwards - slightly baked. 80 m - slightly jointed. 100 m - slightly jointed + *pyrite* staining. 134-136 m - thin layers jointed grey/green quartzite. 135 m - jointed (blocky ≥ 20 mm). 137 m - jointed, fill *quartz* (apophyllite?), slightly weathered. 142 m - *quartz* + *pyrrhotite* mineralisation. 143 - 146 m - weakly jointed + minor *calcite*.

Data source: Woodford, pers. comm.

BOREHOLE:	**G39973 (obs)**
DRILLER:	Department of Water Affairs and Forestry
COORDINATES:	Latitude, Longitude
DRILLING DATES:	2000
TOTAL DEPTH:	174 m
COLLAR ELEVATION:	
WATER LEVEL (03-02-2000):	102.48 m

Depth (m)	**Lithology**
0-5	Weathered, light grey mudstone with white precipitate
5 - 12	Fresh, light grey, fracture stained mudstone. Water strike at 12 m. Highly baked and fractured mudstone at 12 m. Chippings up to 7 cm in length.
12-29	Fresh, medium grained dolerite, fractured in places.
29 - 32	Light grey mudstone, showing faint signs of brecciation (very small crystals).
32 - 102	Dark grey to black, baked shale, with bands of light grey shale. Very few signs of brecciation.
102 - 174	Light grey mudstone, slightly coarser grained (possibly very fine grained sandstone). 146-148 m & 150-174 m - medium scale brecciation. The mudstones are brecciated, with apophyllite (and other) crystals, from 146 - 174 m.

Data source: R Murray.

APPENDIX 2.2:

Letter to Calvinia's Town Engineer on borehole G39973 as a domestic supply borehole

CSIR
ENVIRONMENTEK

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Mr EJ Wentzel
Private Bag X14
Calvinia
8190

10 December 1998

Re: Domestic water supply using borehole G 39973 (in the artificial recharge plug)

Dear EJ

With the pump in at it's current depth of about 127 m you will be able to abstract approximately 24 000 kL before the water level in the borehole reaches the pump. This is *at least* a half to a third of the available water that is stored in the underground reservoir.

In terms of utilising this water at the moment, that is prior to any blending within the plug, *we recommend that you mix this water with dam water at a ratio of 1 : 8* (that is, 1 litre plug water with 8 litres dam water).

This is necessary because the plug's water, in it's natural form, that is, prior to blending, contains certain elements in concentrations which are above the recommended drinking water guidelines. Once we start to store other water (dam and borehole) inside the plug, we expect the concentrations of these elements to go down. So initially, when you use "pure" plug water, you will need to dilute it by mixing it with dam water.

Please note that the pump in borehole G 39973 (at it's current depth of 127 m) is designed to deliver 7.5 L/s or 27 kL/hr to the reservoir, and about 7.0 L/s to the dam (J Van Schalkwyk, Wouter Engelbrecht Inc, personal communication). Should you require 1 000 kL/day, and assuming the pump delivers 7.5 L/s you should operate the pump for 4 hours a day (± 110 kL/day) in order to get a 1 : 8 mix. Operating at this rate it will take about 220 days, or more (7 months) to reach the pump intake.

Please note that you should pump the plug water to the dam, and not the mixing chamber before the reservoir. The reason for this is that the concentrations of the elements of concern may decrease due to the oxidising conditions in the dam and/or the modified chemical equilibrium after dilution with the dam water.

We recommend that you initially *sample the water every week* in order to: i) establish the changing water quality of the plug with pumping, and ii) the blending of water in the dam.

In order to do so, you will need to collect:

i) Two 250 mL samples of plug water in plastic bottles (the ones I left with you), after pumping the borehole for four hours.

ii) Two 250 mL samples of blended water at the reservoir outlet (after treatment) in plastic bottles.

Send them to the people below, and have them analysed for the following:

Aluminium, arsenic, boron, lead & mercury (One sample from the plug, and one at the reservoir after blending)	*Darrel Whyte, Environmentek, CSIR, Meiring Naude Street, Brummeria, 0002, Pretoria*
Fluoride and iron (One sample from the plug, and one at the reservoir after blending)	*Mike Louw, Environmentek, CSIR, Jan Cilliers Street, Stellenbosch*

It is important to get the sample to them as soon as possible after collection; for this reason, please courier them. After a few weeks, we will be in a position to establish how frequently the sampling should be over the long term.

The first table in the attachment lists the elements which are above the recommended guidelines for drinking water, and their concentrations. Information on them is provided in the following pages. The table also gives the concentrations after the water has been mixed at a ratio of 1 : 8, and the risk group they will fall into after mixing at 1 : 8.

Please also note the following positive factors which indicate that the water quality is likely to improve:

i. By mixing water in the dam, the ratios are likely to be far greater than 1 : 8.
ii. The water quality tends to improve with depth (ie pumping). See column two of the first table - after 7.8 days of pumping the concentrations of arsenic, lead and mercury decreased.
iii. The water quality tends to improve after a substantial volume has been removed. Compare arsenic and lead in columns two (1994) to those in column four (1998).

We also expect the water quality to improve as we inject dam (and possibly groundwater) into the plug, and therefore the blending ratio of 1 : 8 will likely increase in future. In order to establish the plug's water quality in the long term, we need to empty it as soon as possible and fill it with dam and borehole water. We must aim to do this several times over the next two years so that by the time our Water Research Commission project is complete in 2001, you have a good idea of how to manage this sub-surface storage system.

Yours sincerely

Ricky Murray

Attachment to letter dated 10 December 1998: Domestic water supply using borehole G 39973 (in the artificial recharge plug)

Borehole G 39973

Determinand	*10/10/94*[1] *18/10/94*[2]	*26/05/98*	*18/08/98*	*Concentrations after the water has been mixed at a ratio of 1 : 8*	*Risk group after mixing at 1 : 8*
Aluminium as Al mg/L		0.17		0.02	Target water quality range[4]
Arsenic as As mg/L	0.412 0.311		0.325	0.039 - 0.051	Rare instances of negative effects[5]
Boron as B mg/L		0.8		0.1	Recommended limit[6]
Fluoride as F mg/L		11.5	11.9	1.4 - 1.5	Rare instances of negative effects[5]
Iron as Fe mg/L		0.14		0.02	Target water quality range[4]
Lead as Pb mg/L	0.035 0.0196		0.0014	0.0002 - 0.0044	Target water quality range[4]
Mercury as Hg mg/L	0.048 0.0109			0.0014 - 0.006	Maximum permissible limit[6]

[1] At the start of the 7.8 day constant discharge test

[2] At the end of the 7.8 day constant discharge test

[4] Department of Water Affairs and Forestry (1996)

[5] Department of Water Affairs and Forestry and Department of Health (1996)

[6] Kempster and Smith (1985)

Comments on the elements which, when mixed at a ratio of 1: 8, are below the "Target water quality range" or the "Recommended limit":

Aluminium	The value of 0.17 mg/L is slightly above the "Recommended limit" of 0.15 mg/L as described by Kempster and Smith (1985). The "Maximum permissable limit" is given as 0.5 mg/L and the "Crisis limit" as 1.0 mg/L. The "target range" for aluminium is 0 - 0.15 mg/L (DWAF, 1996).
Boron	The value of 0.8 mg/L is slightly above the "Recommended limit" of 0.5 mg/L as described by Kempster and Smith (1985). The "Maximum permissable limit" is given as 2.0 mg/L and the "Crisis limit" as 4.0 mg/L.
Iron	The value of 0.14 mg/L is slightly above the "Recommended limit" of 0.1 mg/L as described by Kempster and Smith (1985). The "Maximum permissable limit" is given as 1.0 mg/L and the "Crisis limit" as 2.0 mg/L. The "target range" for iron is 0 - 0.1 mg/L (DWAF, 1996).
Lead	The value of 35 μg/L is below the "Recommended limit" of 50 μg/L mg/L as described by Kempster and Smith (1985), but above the "target range" for lead, which is 0 - 10 μg/L (DWAF, 1996).

Health effects for arsenic, fluoride and mercury

The following tables give the specifications for:

- the "guide for the health related assessment of the quality of water supplies" (Department of Water Affairs and Forestry and Department of Health, 1996);
- the "guidelines" for drinking water quality (World Health Organisation, 1996 and Department of Water Affairs and Forestry, 1996);
- the "proposed drinking-water criteria" (Kempster and Smith, 1985).

Department of Water Affairs and Forestry and Department of Health (1996):

A guide for the health related assessment of the quality of water supplies

Row No	*Arsenic as As μg/L*		*Fluoride as F mg/L*	
1	0 - 10	No health effects	0 - 1.0	No health effects
2	10 - 50	Slight risk of health effects.	1.0 - 1.5	Slight mottling of teeth in sensitive individuals.
3	50 - 200	Chronic effects in sensitive individuals.	1.5 - 3.5	Noticeable mottling of teeth in most continuous users. Pitting of teeth enamel.
4	200 - 2000	Chronic arsenic poisoning with long term intake.	> 3.5	Skeletal fluorosis occurs on long-term exposure.

The Department of Water Affairs and Forestry and Department of Health document says the following in relation to these limits:

Row 1	Suitable for use
Row 2	Rare instances of negative effects
Row 3	Common instances of negative effects
Row 4	Unsuitable for use without treatment

Department of Water Affairs and Forestry (1996): South African Water Quality Guidelines

Effects of Arsenic on Human Health

Arsenic Range ($\mu g/\ell$)	*Effects*
Target Water Quality Range 0-10	*No health effect expected; ideal concentration range.*
10 - 200	Tolerable concentration, but low risk of skin cancer in highly sensitive individuals over long term.
200 - 300	Increasing possibility of mild skin lesions over long term. Slight possibility of induction of skin cancer over long term.
300 - 600	Possible adverse, chronic effects in sensitive individuals; brief exposure has no effect; skin lesions, including hyper pigmentation will begin to appear on long-term exposure.

Note: It is recommended that the concentration of arsenic in potable water should never exceed 200 $\mu g/\ell$, but ideally should not exceed the Target Water Quality Range (10 $\mu g/\ell$).

Effects of Fluoride on Aesthetics and Human Health

Fluoride Range (mg/ℓ)	*Effects*
Target Water Quality Range *0 - 1.0*	*The concentration in water necessary to meet requirements for healthy tooth structure is a function of daily intake and hence varies with annual maximum daily air temperature. A concentration of approximately 0.75 mg/ℓ corresponds to a maximum daily temperature of approximately 26 °C - 28 °C. No adverse health effects or tooth damage occurs.*
1.0 - 1.5	Slight mottling of dental enamel may occur in sensitive individuals. No other health effects are expected.
1.5 - 3.5	*The threshold for marked dental mottling with associated tooth damage due to softening of enamel is 1.5 mg/ℓ.* Above this, mottling and tooth damage will probably be noticeable in most continuous users of the water. No other health effects occur.
3.5 - 4.0	Sever tooth damage especially to infants' temporary and permanent teeth; softening of the enamel and dentine will occur on continuous use of water. *Threshold for chronic effects of fluoride exposure*, manifested as skeletal effects. Effects at this concentration are detected mainly by radiological examination, rather than overt.
4.0 - 6.0	Severe tooth damage especially to the temporary and permanent teeth of infants; softening of the enamel and dentine will occur on continuous use of water. Skeletal fluorosis occurs on long-term exposure.
6.0 - 8.0	Severs tooth damage as above. Pronounced skeletal fluorosis occurs on long-term exposure.
> 8.0	Severe tooth damage as above. Crippling skeletal fluorosis is likely to appear on long-term exposure.
> 100	Threshold for onset of acute fluoride poisoning, marked by vomiting and diarrhoea.

Note: *It is recommended that the concentration of fluoride in potable water never exceed 4.0 mg/ℓ, due to the likelihood of skeletal fluorosis with crippling, as well as the loss of teeth.*

Effects of Mercury on Human Health

Mercury Range (µg/ℓ)	*Effects*
Target Water Quality Range 0 - 1	*No health effects expected*
1 - 5	At two µg/ℓ water provides up to 13% of the safe daily intake, and at five µg/ℓ water provides 33 % of the safe daily intake. Risk to the general population is unlikely. A very slight risk of neurotoxicity due to organic mercury may exist for sensitive individuals at the upper limit of this range.
5 - 20	Brief, episodic exposure is unlikely to have adverse effects. Some risk of chronic neurotoxic effects due to organically complexed mercury.
20 - 50	Risk of neurotoxicity with consequent serious disability with continuous exposure.
50 - 1 000	Risk of neurotoxicity with continuous exposure, particularly to organic mercury compounds. Acute effects may occur, particularly with organic mercury compounds.
1 000 - 10 000	Chronic effects; acute poisoning with damage to the nervous system and brain may occur with single exposure, and definitely with continuous exposure.
> 100 000	Acute poisoning; permanent brain damage and/or death is possible.

World Health Organisation (1996):

Guidelines for drinking-water quality

Determinand	***Provisional guideline value***	***Comments***
Arsenic as As µg/L	10	The estimated excess life-time risk of skin cancer associated with exposure to this concentration is 0.0006 (That is, 1 in 60 000 people could get skin cancer if exposed to this concentration for a life-time).
Determinand	Guideline value	Comments
Fluoride as F mg/L	1.5	Concentrations above this value carry an increasing risk of dental fluorosis, and much higher concentrations lead to skeletal fluorosis.
Mercury as Hg µg/L	1.0	In deriving this value, it was assumed that 10% of the "provisional tolerable weekly intake" comes from water.

Kempster and Smith (1985):

Proposed aesthetic/physical and inorganic drinking-water criteria for the Republic of South Africa

Determinand	*Recommended limit*	*Maximum permissible limit*	*Crisis limit*	*Effects other than aesthetic*
Arsenic as As µg/L	100	300	600	Nutritionally essential in small quantities. Toxic in excess.
Fluoride as F mg/L	1.0	1.5	3.0	Related to incidence of caries.
Mercury as Hg µg/L	5	10	20	Organic complexes neurotoxic

The Kempster and Smith document says the following in relation to these limits:

The *recommended* or working limit is the limit which should ideally not be exceeded. The recommended limit has a built-in safety factor, and thus no immediate danger exists where this limit is exceeded, provided the maximum permissible limit is not exceeded.

The *maximum permissible limit* is still safe, but should not be exceeded. Where the concentration of a particular determinant exceeds the maximum permissible limit, then planning/action to reduce the concentration of this pollutant should be instituted without delay.

In addition, it is suggested that a limit be set on the amount by which the concentration of a determinant may exceed the maximum permissible limit before extreme action need to be taken. This *crisis limit* was originally defined as twice the maximum permissible concentration limits.

In applying these criteria, the crisis limit should be treated as a tentative guideline only, and not applied rigidly, except in the case of extremely toxic determinants, such as cyanide, where the risk associated with elevated concentrations is high. For the aesthetic determinants, as well as for determinants of low toxicity, where there is only a slight risk at elevated concentrations, the crisis limit should be used with discretion and may be relaxed where circumstances warrant.

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Department of Water Affairs and Forestry, 1996. South African Water Quality Guidelines. Volume 2. Domestic Use.

Department of Water Affairs and Forestry and Department of Health, 1996. A guide for the health related assessment of the quality of water supplies. First Edition.

Kempster, P. L. and Smith, R., 1985. Proposed aesthetic/physical and inorganic drinking-water criteria for the Republic of South Africa. CSIR Report No. 628.

SABS 241-1984. Specification for Water for Domestic Supplies. SABS, Pretoria.

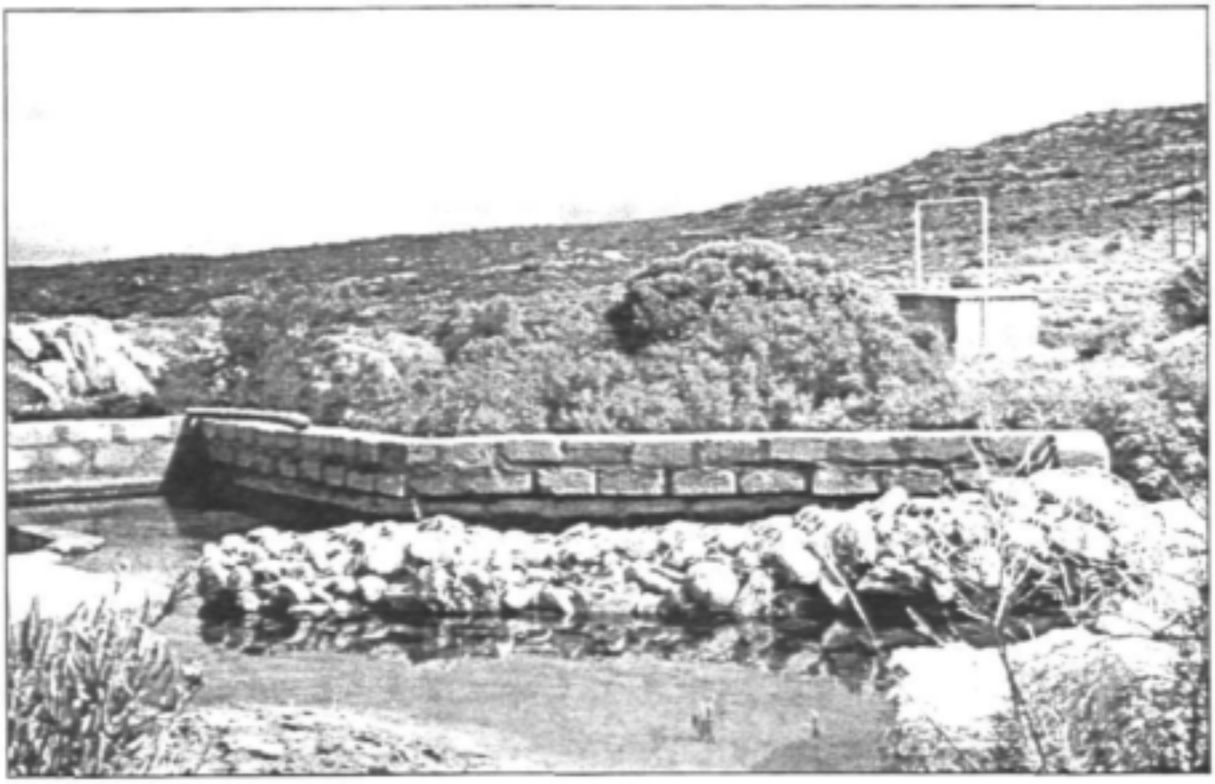
World Health Organisation, 1996. Guidelines for drinking-water quality. Second Edition. Geneva.

SECTION 3:

Karkams Artificial Recharge Study

by

E C Murray & G Tredoux



The sand filter where water is filtered before it gravitates as injection water into borehole G45757 or G39002. In the background is the pump house of abstraction borehole G39002.

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1. INTRODUCTION

The village of Karkams (Plate 1), with a mean annual rainfall of 250 mm, and a population of 1690, depends solely on groundwater. The town's average consumption of 225 m^3/day (2000 and 2001) is high because the town hosts a large boarding school with waterborne sewerage.

The town is supported by three boreholes, of which G39002, the borehole used in the injection tests, is the lowest yielding. The average abstraction from this borehole since March 1995, when records were kept, is 284 m^3/month or 9.5 m^3/day. This borehole was chosen for the artificial recharge study because it had previously been used for injection. Artificial recharge was initially recommended at this site by DWAF (Esterhuyse, 1990). Water level, water quality and abstraction had also been monitored by Toens and Partners since 1995 when the borehole was equipped for production (Toens & Partners, 1997).

Although no detailed monitoring of the artificial recharge process had taken place, the infrastructure for recharge was in a reasonable condition, and only a few changes were required in order to make it suitable for a more detailed study of the borehole injection process. The changes included re-designing the sand filter, installing a flow meter, drilling a new injection borehole and drilling new observation boreholes.

Historic water level data from borehole G39002 indicates that the sustainable yield of the borehole is 6.5 m^3/day, even though the borehole's drilling (or "blow") yield was 25.9 m^3/h. Relatively high drilling yields, such as that found in borehole G39002, can be obtained in this area by siting boreholes on faults, however, the low natural recharge and limited size of some of the groundwater compartments in the area limits their sustainable yields.

The aims of artificially recharging the Karkams aquifer is to rapidly replenish the aquifer when surface runoff is available and to improve the groundwater quality.

Three controlled injection tests were conducted with the upgraded infrastructure:

1st injection test:	17 September to 29 October, 1999 1 042 m^3 injected into existing production borehole, G39002
2nd injection test:	15 July to 13 October, 2000 4 162 m^3 injected into new injection borehole G45757
3rd injection test:	6 July to 21 November, 2001 6 567 m^3 injected into both G45757 and G39002

The layout of the scheme is presented in Figure 6.

Plate 1 Karkams village with the boarding school on the right hand side of the picture

2. THE HYDROGEOLOGY OF THE KARKAMS AQUIFER

The Karkams aquifer consists of porphyritic granites, on which borehole G39002 is located, and older, fine-grained granitic gneisses. Most of the high yielding boreholes have been sited on the north-south striking faults that cut through this mountainous area (Figure1).

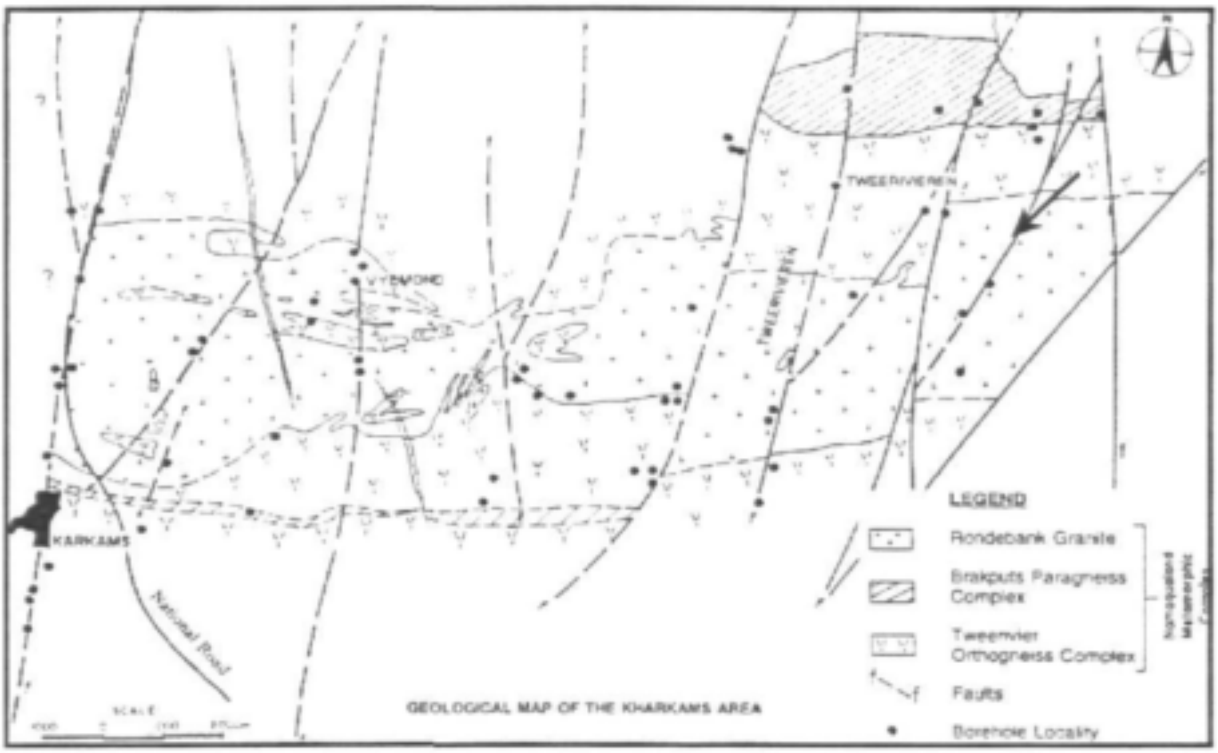

Figure 1 Geology of the Karkams area showing major faults and borehole locations - the arrow points to borehole G39002 (after Visser, *et al*, 1995)

Groundwater levels are generally a subdued replica of topography, with borehole G39002 being located in a high lying area close to the natural recharge area (Figure 2).

The groundwater salinity in the region generally increases from east to west (Figure 3 & 4) and is directly related to elevation above sea level, which in turn, controls precipitation (Visser, *et al*, 1995).

The fluoride content of the groundwater in the area is generally high which is not uncommon in granitic and gneissic terrains. The trend generally follows that of salinity, with the highest values found in the low lying areas in the west (Visser, *et al*, 1995). The fluoride concentration of 2.8 mg/L at borehole G39002 is relatively low in comparison to some boreholes in the west which have values up to 5.3 mg/L (Figure 4 & 5).

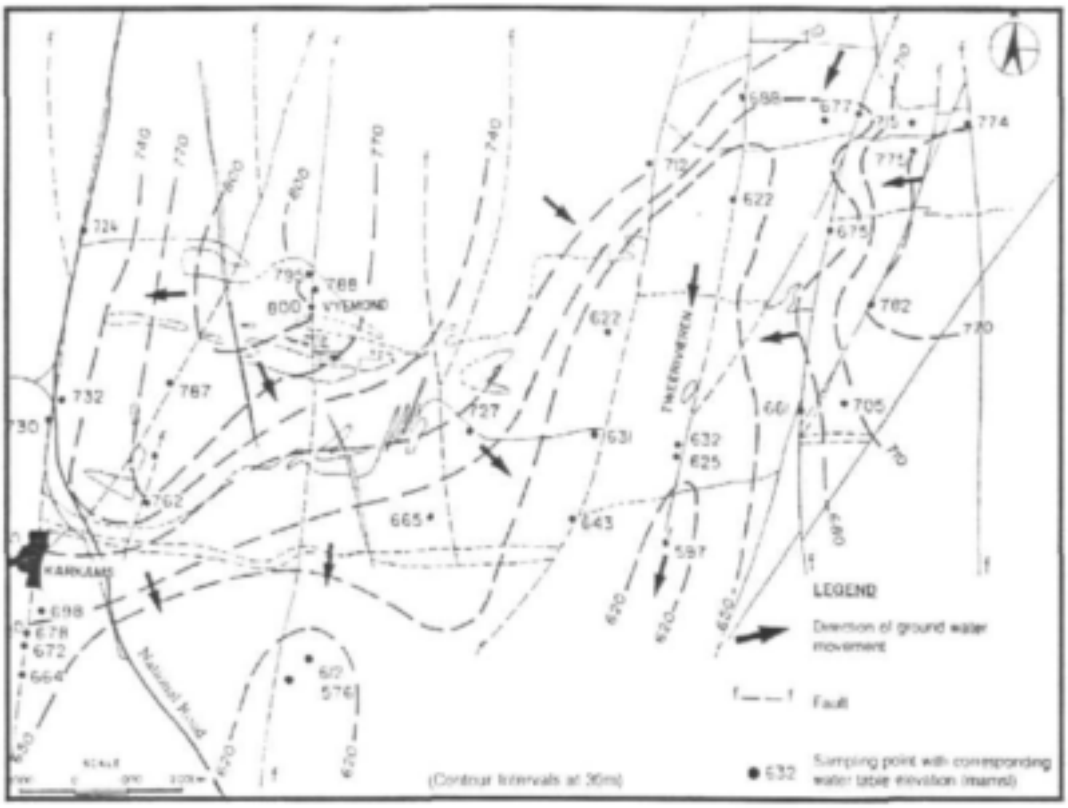

Figure 2 Contour map of groundwater levels (mamsl) in the Karkams area (after Visser, *et al*, 1995).

Figure 3 Contour map of groundwater salinity (after Visser, *et al*, 1995)

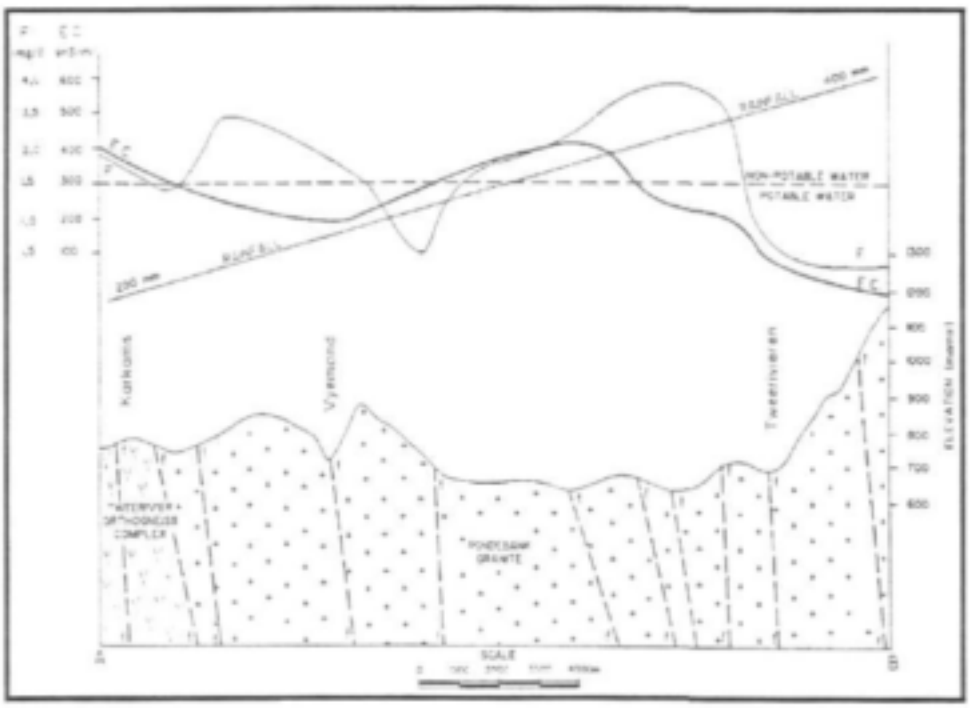

Figure 4 Geological section showing electrical conductivity and fluoride content of groundwater (after Visser, *et al*, 1995)

Figure 5 Contour map of fluoride content in groundwater (after Visser, *et al*, 1995)

3. LAYOUT OF THE ARTIFICIAL RECHARGE SCHEME

3.1 Borehole layout

Figure 6 shows the layout of the artificial recharge scheme. All boreholes except for G45755 intercept fractured granites associated with a NNW-SSW fault. Abstraction borehole G39002, with an average production rate of 9.5 m^3/day, is equipped with a submersible pump. This borehole was used for injection in 1995 and 1996 prior to the CSIR's involvement in this project. During these injection runs, water was gravitated at a high rate into the borehole, and the water level in the borehole rose rapidly to above ground level and overflowed into the pump house. The filter was reported to be faulty, and sand was entering the borehole.

Prior to injection in 1999, the sand filter was re-designed, a flow meter installed in the injection pipeline and a piezometer tube installed into the borehole. Four observation boreholes, G45755 - G45758, were drilled by DWAF shortly before the rains came in 1999.

After completion of the 1999 injection run, borehole G45757 was converted into an injection borehole (Plate 2). The main reason for this was because borehole G39002 is a small diameter hole (127 mm PVC), and it was not possible to equip it with an injection pipe. Water entering the borehole would thus cascade down the hole. This raised the concern that excess air could enter the aquifer, and that this could lead to air entrapment and undesirable chemical or microbial reactions.

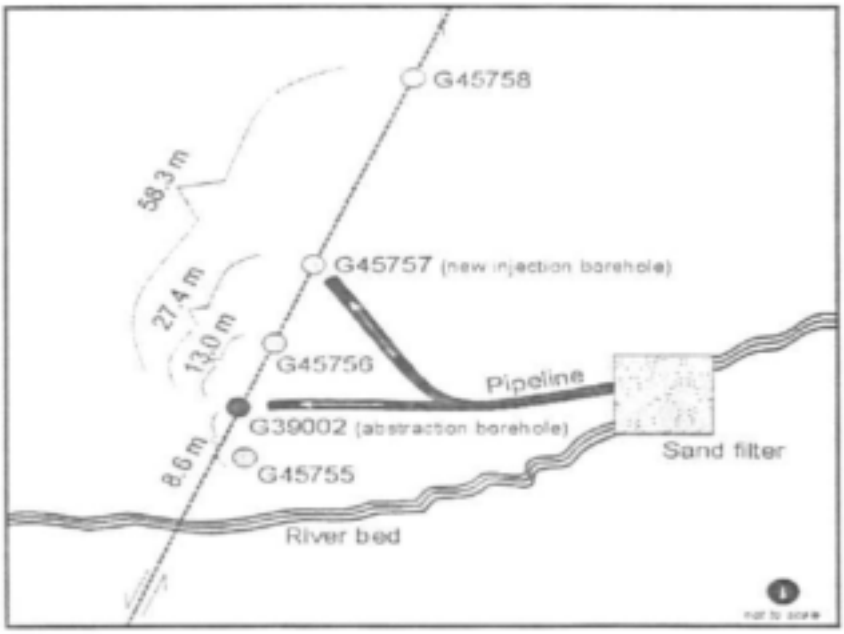

Figure 6 Layout of the borehole injection site (natural groundwater flow direction is from East to west (left to right in this sketch)

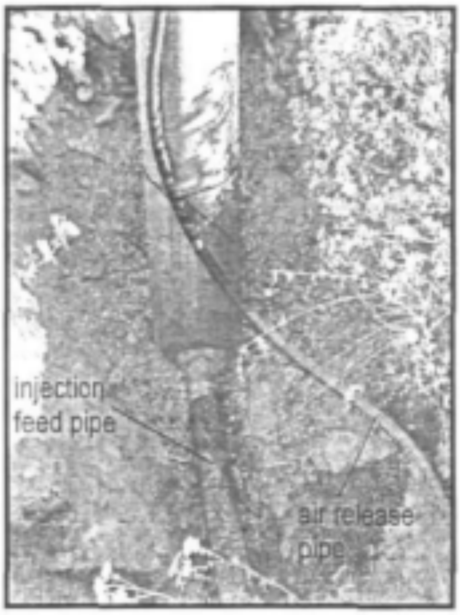

Plate 2 New injection borehole G45757. The injection feed pipe from the sand filter enters the borehole below ground level so that water can flow by gravity into the borehole. Layflat hose piping is attached to the injection feed pipe on the inside of the borehole.

3.2 Filter design

Mr H. A. De Villiers re-designed the filter (Appendix 3.1). The main new features were: an appropriately sized screen at the base of the filter; correctly sized filter sand, grading from coarse at the base to relatively fine at the surface; a heightened filter wall; and gabions at the entrance to the filter to prevent debris from entering the filter. The filter was designed to give a flow of 0.5 L/s. Plates 3 - 7 show some of these features.

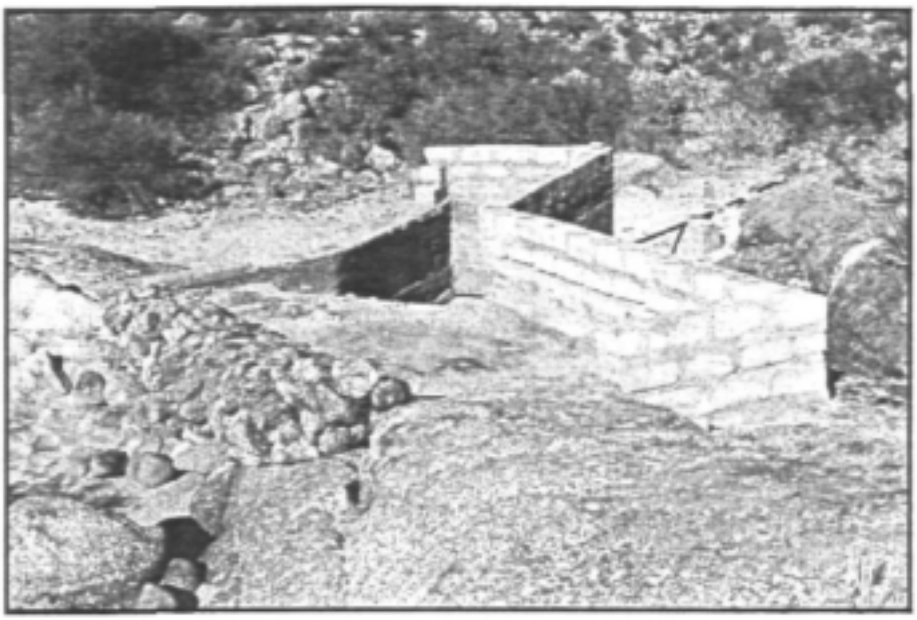

Plate 3 The sand filter before it had been filled with sand

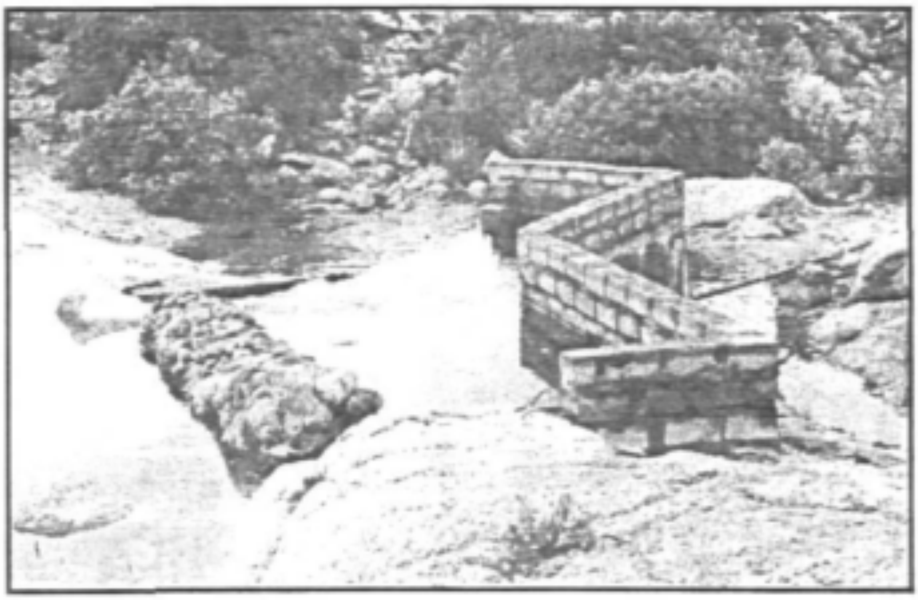

Plate 4 The sand filter during operation

Plate 5 Slotted PVC filter pipe at base of the sand filter. This is surrounded by coarse gravel (5 - 10 mm diameter), which is covered by 16/40 pool filter sand to 10 cm above the gravel, and then covered with 2mm sieved river sand to the top of the filter.

Plate 6 2mm sieved river sand. The sand < 2mm in diameter is discarded.

Plate 7 The filter ready for operation, showing the new gabion, the new wall and filtered river sand covering the surface of the filter.

4. HYDROGEOLOGY OF THE ARTIFICIAL RECHARGE AREA

4.1 Geological logs

A cross section of the boreholes is presented in Figure 7. The geology can be summarised as having a thin layer of weathered granite (up to a maximum depth of 10 m), followed by fresh granites with fractures in places. The geological logs of each borehole are summarised in Appendix 3.2.

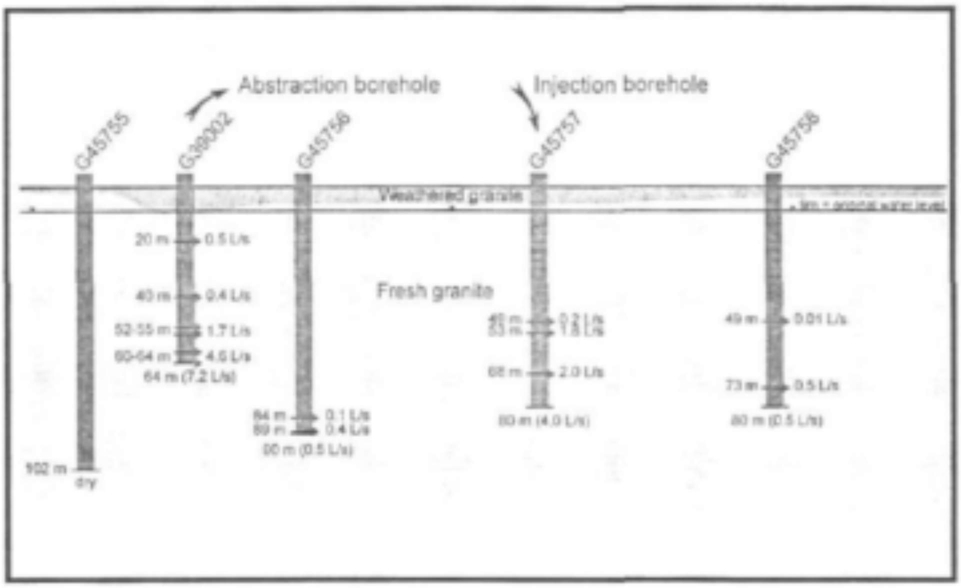

Figure 7 A cross section of the boreholes

4.2 Geophysical logs

The newly drilled boreholes were geophysically logged by DWAF. Figure 8 presents the one set of density logs. The full set of geophysical logs are presented in Appendix 3.3.

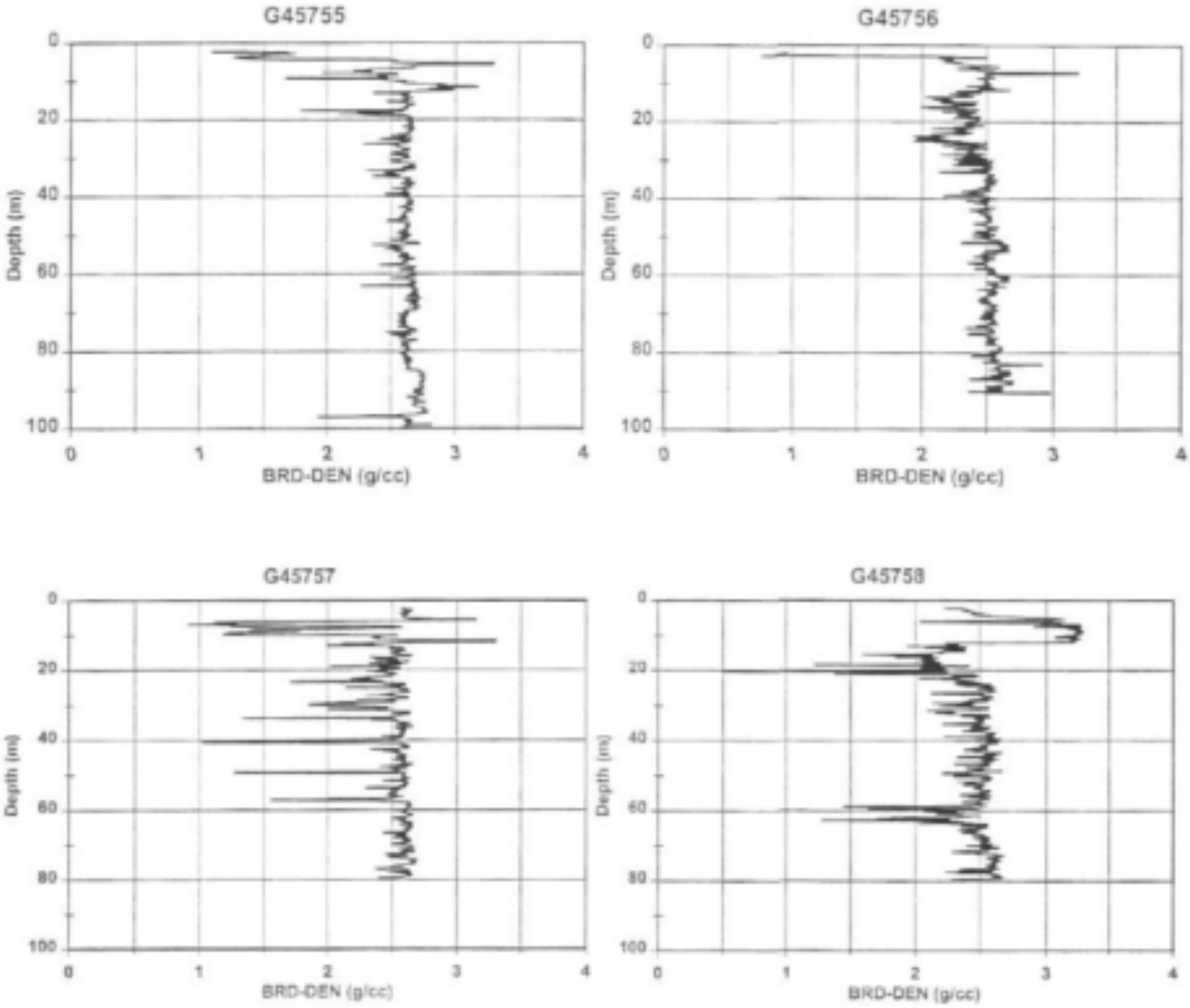

Figure 8 Density logs of the newly drilled boreholes

Figure 8 shows that borehole G45757 intercepted a number of water strikes. For this reason, and because injection water from the sand filter could flow under gravity into this borehole, it was selected as the new injection borehole.

4.3 Geochemistry

4.3.1 Water quality profiles

The groundwater quality at the artificial recharge site deteriorates with depth. Salinity profiles in the three boreholes located within 30 m from G39002, show that the electrical conductivity (EC) values increase on average from 111 mS/m at 40 mbgl to 256 mS/m at 80 mbgl (Figure 9). The exception is borehole G45758, located 58 m from the abstraction borehole. Here the near-surface salinity is in the order of 350 mS/m, and this increases to 490 mS/m at a depth of 75 m, before dropping to 360 mS/m at 80 m. It is quite remarkable that the salinity of groundwater can differ by such a vast amount over tens of metres. This borehole certainly appears to tap different groundwater to the other observation boreholes.

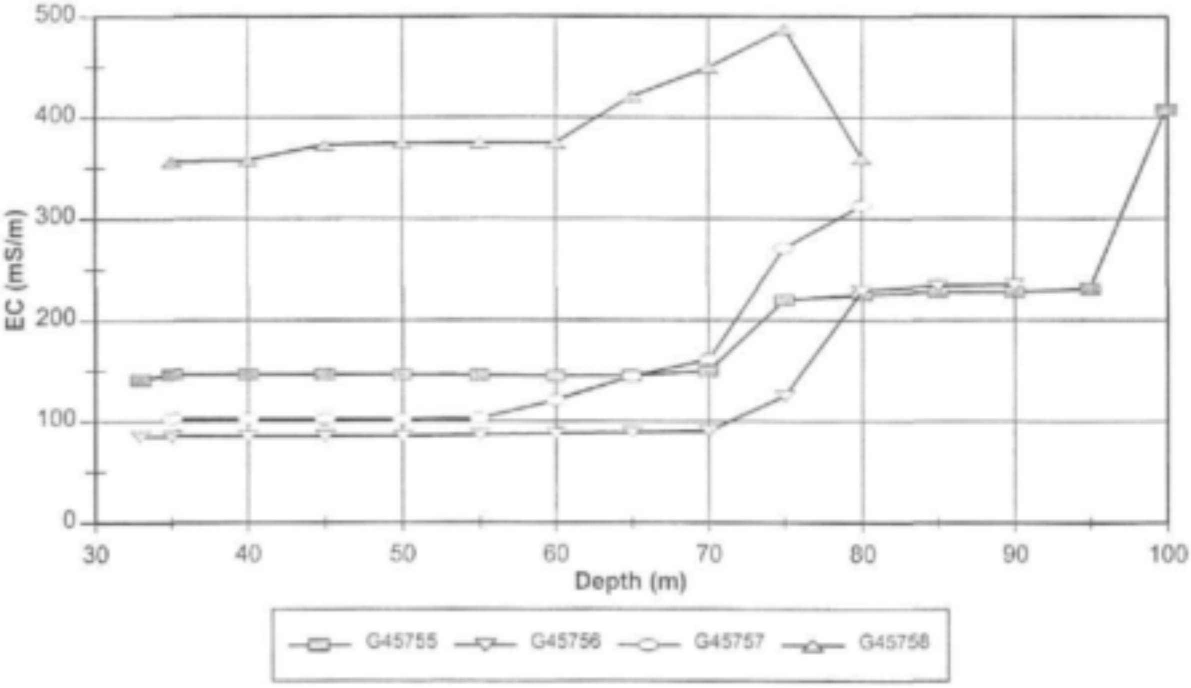

Figure 9 Groundwater salinity profile at the artificial recharge site

Figure 10 presents EC and pH profiles for each borehole. The pH values tend to mirror the EC values - as the EC value increases with depth, so the pH decreases.

It appears that after a depth of about 60 m to 70 m the water quality changes from the relatively young or fresh water to older, more stagnant water. This may explain an interesting phenomenon that has been observed by some locals in Namaqualand: prior to boreholes having been drilled, the taste of water from springs was "good", however, ever since there has been an increase in abstraction from boreholes, the water has become more saline.

Possibly the younger, fresher water was being discharged from springs, and now the deeper, "mixed" water is being abstracted from depth by borehole pumps.

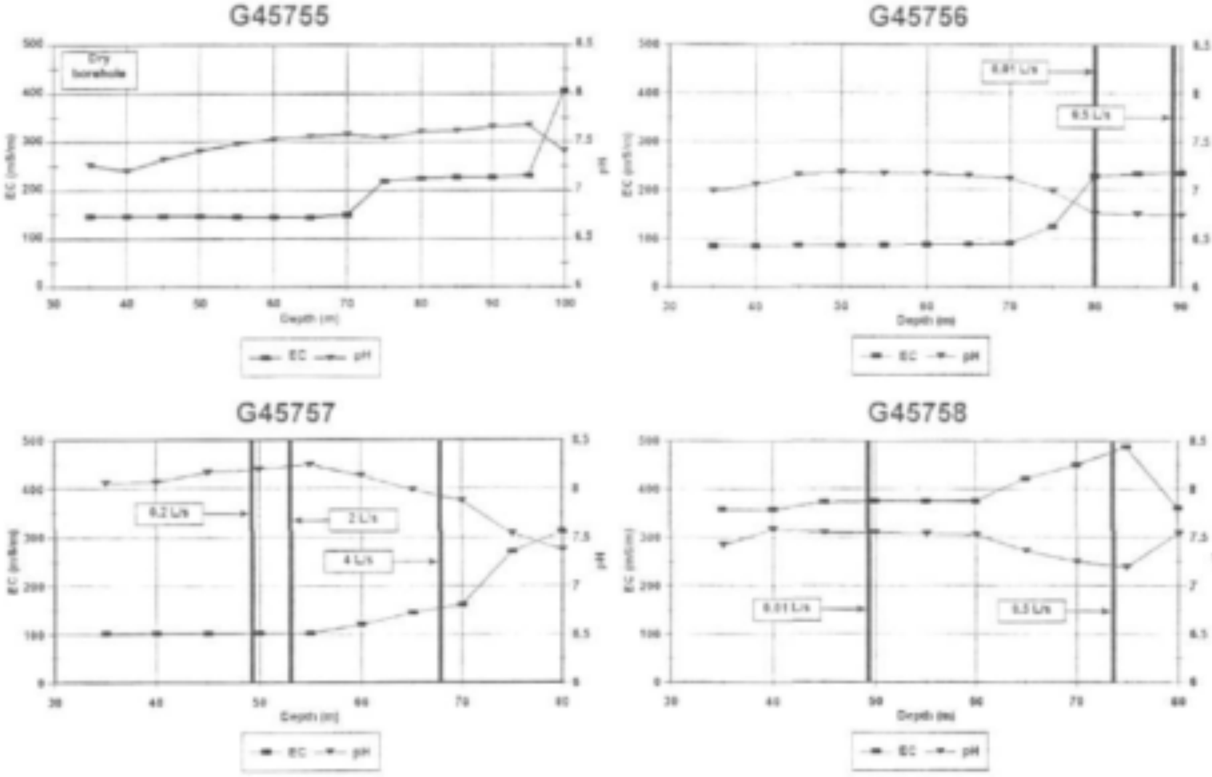

Figure 10 Electrical conductivity and pH profiles of the newly drilled boreholes. Water strikes and drilling "blow" yields are indicated.

4.3.2 Groundwater chemistry

The groundwater from borehole G39002 has high chloride and fluoride concentrations, and a fairly high sodium concentration (Table 1). The water is hard. By diluting this water with fresh surface runoff, the taste, quality and precipitation potential of the water will improve.

Table 1 Groundwater quality form borehole G39002
(sampled with a pump inlet depth of about 50 m)

Determinand	*1989 (16/08/89)*	*1999 (25/05/99)*
Potassium as K mg/L	5.6	4.8
Sodium as Na mg/L	335	280
Calcium as Ca mg/L	127	122
Magnesium as Mg mg/L	91	84
Sulphate as SO4 mg/L	213	201
Chloride as Cl mg/L	776	680
Alkalinity as CaCO3 mg/L		119
Nitrate plus nitrite as N mg/L	0.3	<0.1
Fluoride as F mg/L	2.8	2.4
Iron as Fe mg/L		0.05
Silica as Si mg/L	19.4	10.8
Dissolved Organic Carbon mg/L		3.0
Conductivity mS/m @25°C	282	263
pH (Lab)	8.0	7.3
Saturation pH (pHs) (20°C)		7.5
Total Dissolved Solids (Calc) mg/L	1726	1683
Hardness as CaCO3 mg/L	692	651

5. HISTORICAL GROUNDWATER LEVEL AND QUALITY DATA FROM INJECTION/ABSTRACTION BOREHOLE G39002

Borehole G39002 was drilled in 1989 and test pumped during the same year at a rate of 3.2 L/s for 50 hours. The rest water level prior to pumping was 9.01 mbgl, and after 197 days after the completion of the pumping test, the water level was 12.10 mbgl, or 3 m short of the initial rest water level. Although natural groundwater level fluctuations are not known, this nevertheless suggests that the borehole taps a limited groundwater compartment.

Water was injected in an uncontrolled manner during 1995 and 1996. The rate of injection was unknown, since there was no flow meter and the water level rapidly rose to above ground level where it flooded the pump house and ran back into the stream. During this period water levels were not accurately measured, however, water salinity in terms of EC was being accurately measured. Figure 11 presents the data that was available prior to the CSIR's involvement.

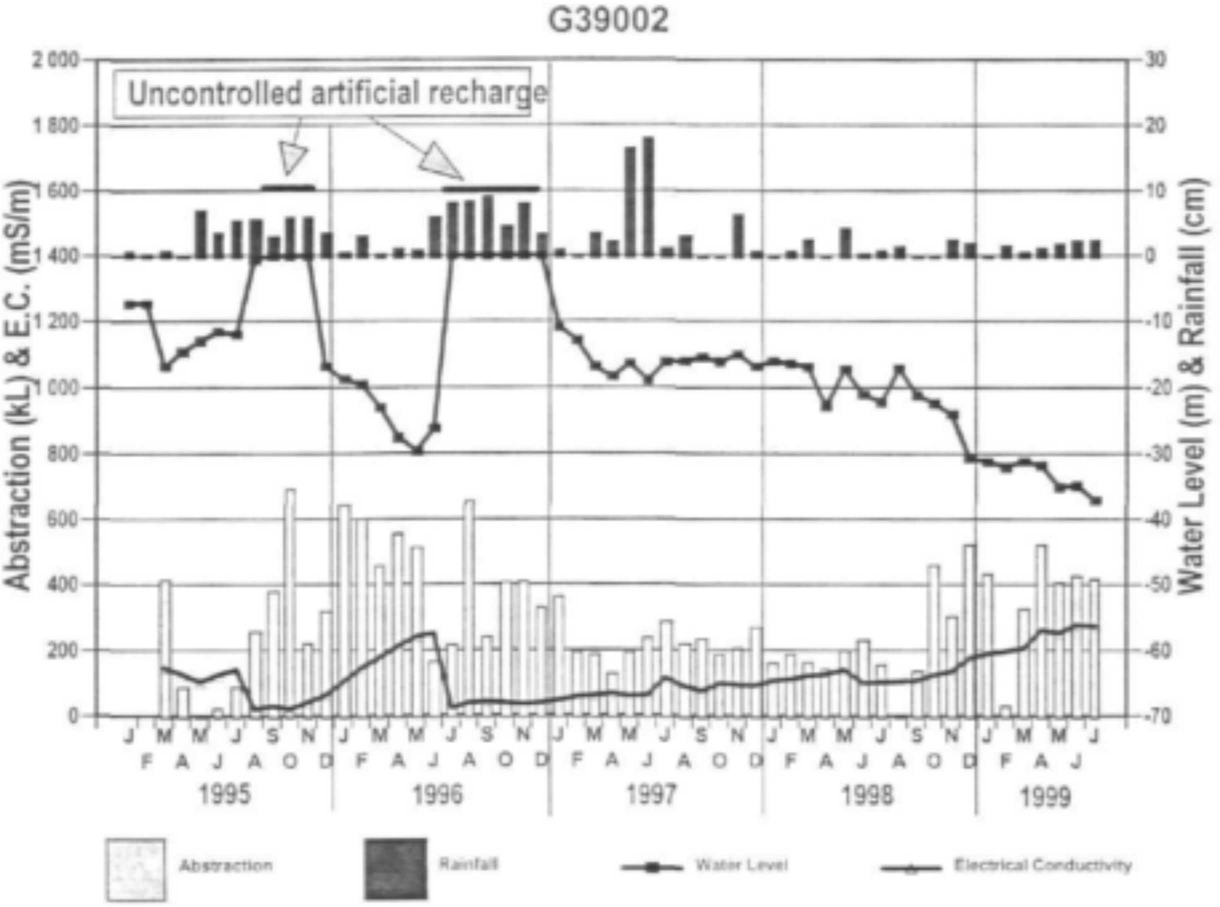

Figure 11 Borehole G39002: Historical abstraction, water level, rainfall and electrical conductivity data (data source: Toens & Partners)

From the historical data the following observations can be made:

Water level

The sustainable yield of the borehole can be considered to be equivalent to the abstraction rate between March 1997 and August 1998, when the water level in the borehole remained relatively flat. The average daily abstraction over this period was 6.5 m^3/day (or 2373 m^3/a).

It is difficult to establish the effect of injection in 1995 on the groundwater level, since during and after the injection period, the borehole was pumped at a relatively high rate. The water level thus dropped rapidly after injection. The long injection period (6 month) during 1996 clearly had a positive impact on the groundwater level. The water level rose by about 10 m (from -30 m to -20 m) even though there was relatively high abstraction from the borehole during this period.

The water level in the borehole dropped rapidly from August 1998 until controlled injection started in October 1999.

Rainfall

In 1996, 525 mm of rainfall was recorded. Most of this (438 mm) fell over a six month period from June to November, and injection ran from July to December. This long injection run had a lasting impact on both groundwater levels and groundwater quality.

Interestingly, groundwater levels and groundwater quality were barely affected by the short, but intense rainfall period in 1997, when 345 mm of rain fell in May and June. No injection took place during this period. The reason for noting this rainfall event will become apparent later when the effect of artificial recharge is discussed.

Water quality

The effect of injection on water quality is shown clearly from the injection runs in 1995 and 1996. Prior to injection in 1995 the EC value was about 120 mS/m. During injection, the water that was sampled was essentially river water, since this water was being fed into the borehole at a rate far greater than the pump's yield. The EC values of about 20 mS/m thus reflects the river's salinity. After this injection run, the water rapidly became saline as groundwater was being abstracted at a high rate. It only took five months for the water to reach an EC value of 200 mS/m.

The 1996 injection run yields far better data. The six months of injection caused the EC value to drop from 250 mS/m down to the river's EC value of 22 mS/m. Thereafter, the EC value rose gently until it reached 292 mS/m in August 1999 prior to controlled injection. After the 1996 injection run, it took twenty seven months for the EC value to rise above the 200 mS/m mark. Of interest, is that the high rainfall months of May and June, 1997, had virtually no effect on the groundwater's salinity.

6. THE CONTROLLED BOREHOLE INJECTION TESTS

Three controlled injection tests were done under the guidance of the CSIR. The first was conducted during the 28 days of streamflow in 1999, where 1 042 m^3 of water was injected into the abstraction borehole G39002; the second was done in the year 2000, where the test was run for 90 days and 4 162 m^3 of water was injected into the newly constructed injection borehole G45757; and the third, in year 2001, ran for 138 days where 6 567 m^3 injected into both G45757 and G39002.

6.1 Water level response

The water level response to injection in all observation boreholes was immediate and the rate of water level rise was the same in all boreholes. The data from borehole G45758, the furthest observation borehole from the abstraction borehole, is presented (Figure 12).

As a result of abstraction in 1999, the groundwater level dropped to 627 meters above mean sea level (mamsl). After injecting 1 042 m^3 in October 1999 and eight months of abstraction thereafter, the groundwater level was 631 mamsl; and after injecting 4 162 m^3 the following year and uninterrupted abstraction, the water level was 633 mamsl in June 2001. If this steady water level rise over the past two years is compared to the steady decline in water levels prior to year 2000 (Figure 11), then it is evident that artificial recharge is having a positive effect on the water level in the aquifer.

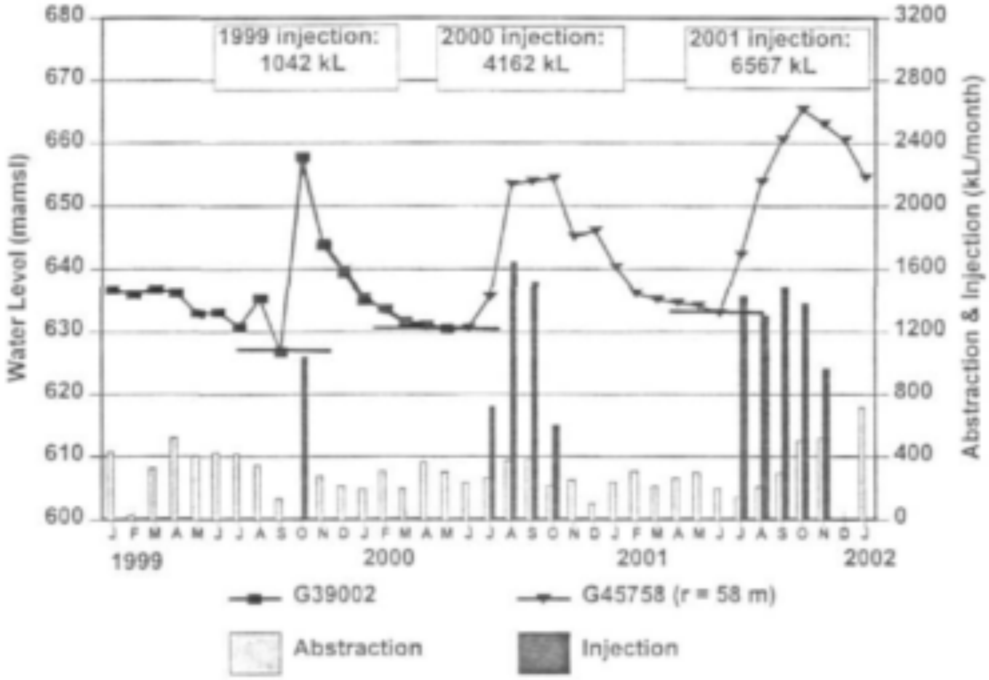

Figure 12 Water level response to controlled artificial recharge

In order to compare the effect of artificial recharge on groundwater levels to that of natural recharge only, Figure 13 has been updated to include the three years where controlled artificial recharge was practiced.

During 1997, 518 mm of rainfall was recorded and 2 681 m^3 was abstracted from borehole G39002. As stated earlier, the water level in the borehole remained relatively stable. During the years 1999 and 2000, the scenario was quite different. In 1999, 228 mm of rainfall was recorded and 3514 m^3 was pumped; and in the year 2000, 240 mm of rainfall was recorded and 3203 m^3 was pumped. During these latter two years the rainfall was about half that of year 1997 and abstraction was considerably more than in 1997, yet the groundwater level appears to have risen following the last two artificial recharge runs.

Had artificial recharge not taken place in 1999 and 2000, it is quite likely that the water levels would have continued to decline, and by May 2001 been closer to 50 mbgl than 36 mbgl.

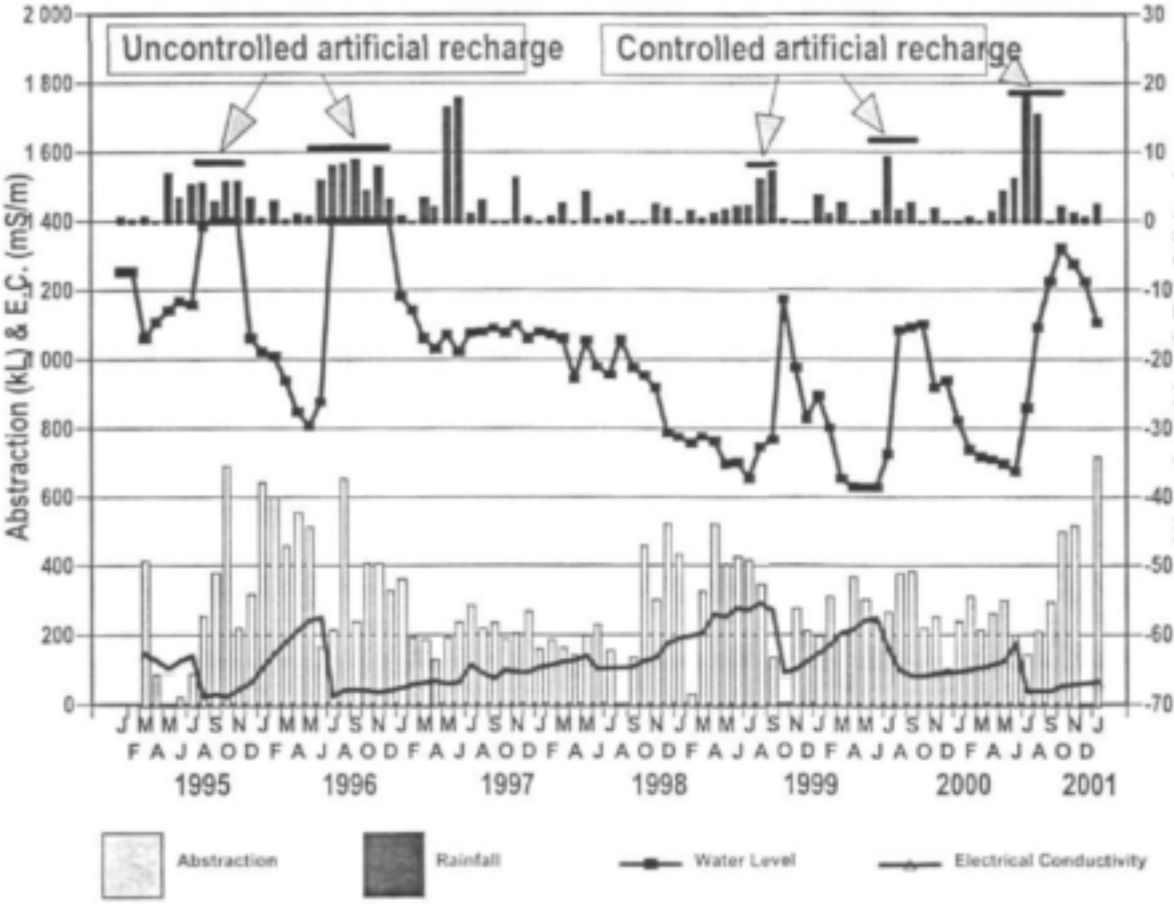

Figure 13 Borehole G39002: Updated abstraction, water level, rainfall and electrical conductivity data

6.2 Water quality response

6.2.1 Water quality response in abstraction borehole G39002

The water quality at the abstraction borehole improved after injecting the filtered river water. During the year 2000 injection run, the average EC value of the river water was 35 mS/m. The EC value of the borehole water dropped from over 250 mS/m before injection in 1999 to less than 100 mS/m after injection in 2000, but this value is increasing as a result of pumping and blending with the natural groundwater (Figure 14).

The salinity response from the controlled injection run in 2000 and 2001 appears to be similar to that of the uncontrolled injection run in 1996. In both cases, the EC values remained relatively low for months after the injection (Figure 13).

In the case of fluoride, the response, as expected, was not as rapid nor as long lasting as that of EC. Figure 15 shows how the fluoride values dropped as a result of injection into borehole G45757 during 2000, but rapidly resumed the ambient values of around 2.6 mg/L. In year 2001, injection was into both the new injection borehole, G45757, and the abstraction borehole, G39002, and for this reason, the fluoride values during injection resemble the river water's values. Unlike the gradual increase in salinity that is seen after injection, the fluoride values are expected to rise to ambient groundwater values within a few months after the completion of the injection run.

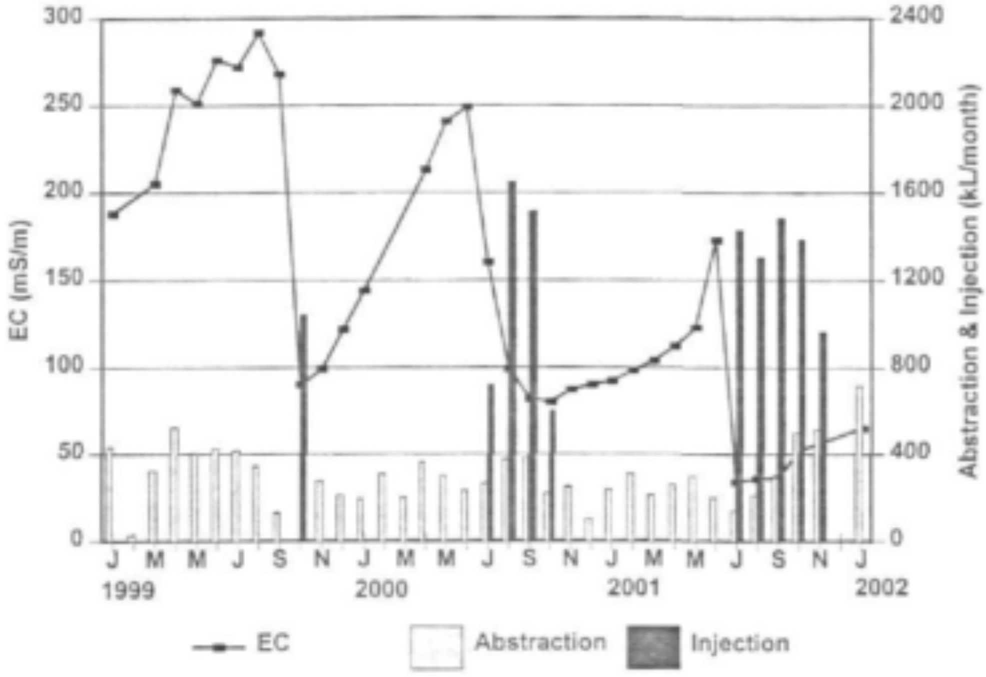

Figure 14 Electrical conductivity response to controlled artificial recharge

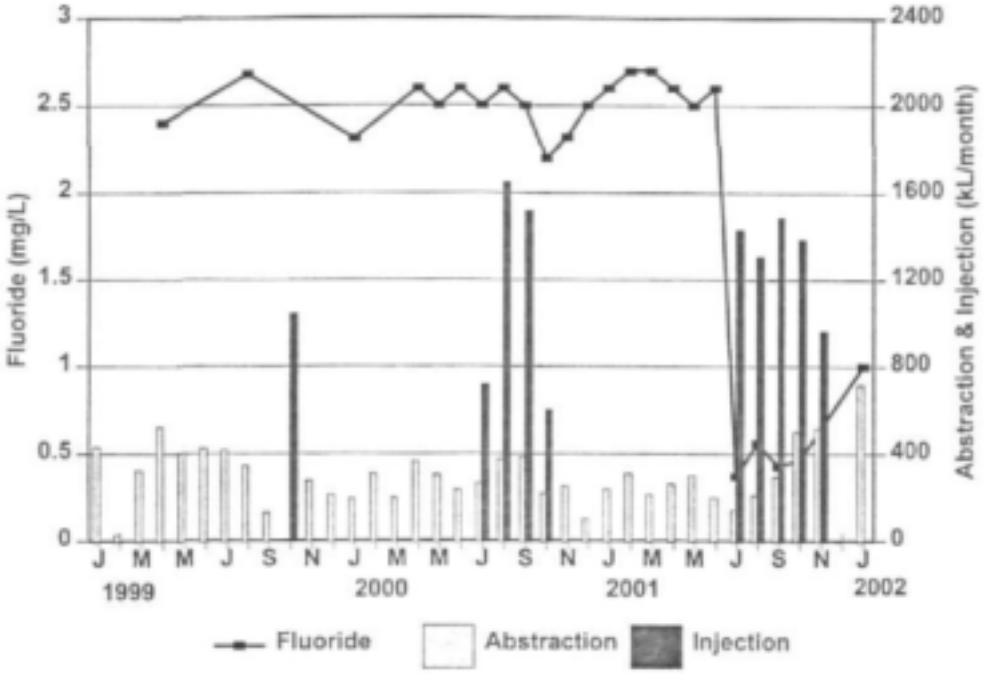

Figure 15 Fluoride response to controlled artificial recharge

6.2.2 Water quality response in observation borehole G45758

From the water quality profiles (Figures 9 & 10), borehole G45758 appears to tap a different groundwater compartment to that of the other observation boreholes. The observation boreholes G45755, G45756 and G45757 have EC values that range between 90 - 150 mS/m over the first 70 m of the borehole, whereas, borehole G45758 has EC values in excess of 350 mS/m over this depth.

A data logger that records EC, pH and water levels was placed in borehole G45758 during the year 2000 injection run. The results show that the water quality in this borehole improved significantly during injection (Figure 16). Prior to the start of injection, the EC value was 380 mS/m (3.8 mS/cm), and during injection the EC value dropped to just over 100 mS/m.

On completion of the injection run, and with abstraction from borehole G39002, the salinity of the water in borehole G45758 began to increase, however, after five months of abstraction, the EC value was still100 mS/m lower (280 mS/m) than prior to injection in July 1999. With repeated injection runs in the future, the water quality of the aquifer can be expected to improve significantly.

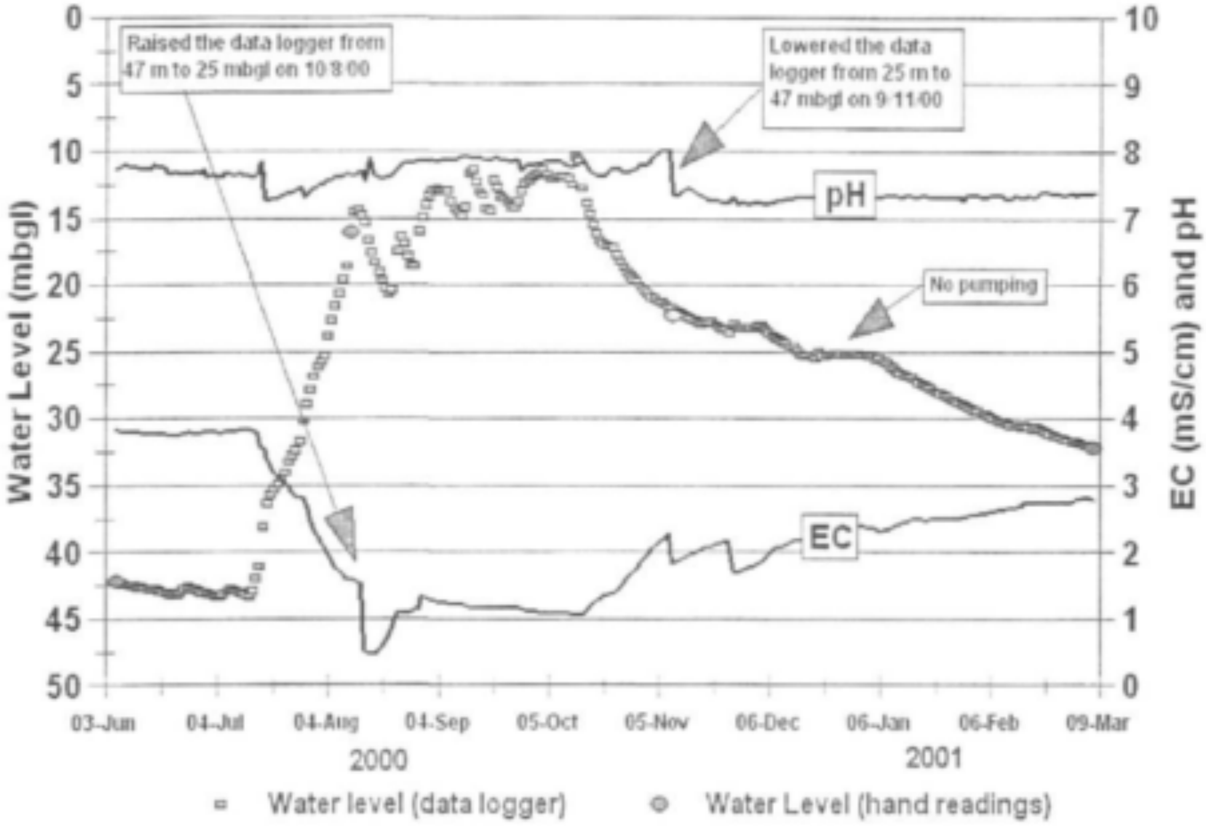

Figure 16 Water quality and water level response in observation borehole G45758 to controlled injection in borehole G45757

The water level fluctuations around the 15 mbgl mark during injection reflect the efficiency of the sand filter. After several days (one to two weeks) the filter would lose efficiency and less water would flow through it. This resulted in the groundwater levels in the boreholes dropping. The top layer of sand in the filter was agitated from time to time in order to remove the fine sand particles that accumulated at the surface. After this, the flow would increase and the water level in the boreholes would rise.

6.2.3 Water quality response with depth

Twenty seven days after the injection run had finished in year 2000, water quality profiles with depth were retaken in boreholes G45755, G45756 and G45758 (Figures 17-19).

Borehole G45755, a "dry" hole, showed that the EC values dropped from ~150 mS/m to ~100 mS/m over the first 65 m; then was the same (~220 mS/m) from 85 - 95 m; and increased to above the pre-injection values towards the bottom of the borehole. It appears as if the injection water seeped into the micro-fractures up to a depth of 65 m. It is difficult to say why the borehole water became more saline towards the bottom of the hole after injection. It does seem like old, stagnant water is held at the the bottom of the borehole.

Borehole G45756, which has it's main water strike at 89 m, gave a uniform EC response after injection. While before injection, the EC values remained constant at ~90 mS/m up to a depth of 70 m, and then rose to ~230 mS/m for the remainder of the borehole, after injection, the EC values were ~50 mS/m throughout the entire hole. The injected water clearly mixed well with the natural groundwater that is available to this boorehole.

Borehole G45757 could not be profiled because of the injection pipeline that was installed in this borehole.

Borehole G45758 (which was discussed in the previous section), has it's main water strike at 73 m, and is generally far more saline than the other boreholes. This borehole responded surprisingly to injection. Prior to injection, the EC values were around 380 mS/m up to a depth of 60 m, after which they rose to close on 500 mS/m at the bottom of the hole. After injection, however, the EC values were relatively constant and less than 200 mS/m throughout most of borehole. This shows that although the borehole appears to tap different groundwater than the other observation boreholes (because of its much higher EC values), it is hydraulically connected to them (or at least connected to the injection borehole).

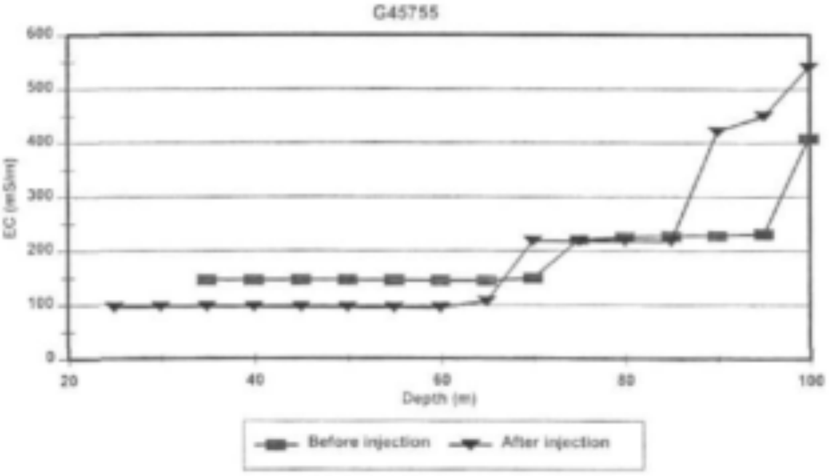

Figure 17 G45755: Electrical conductivity profile before and after injection in 2000

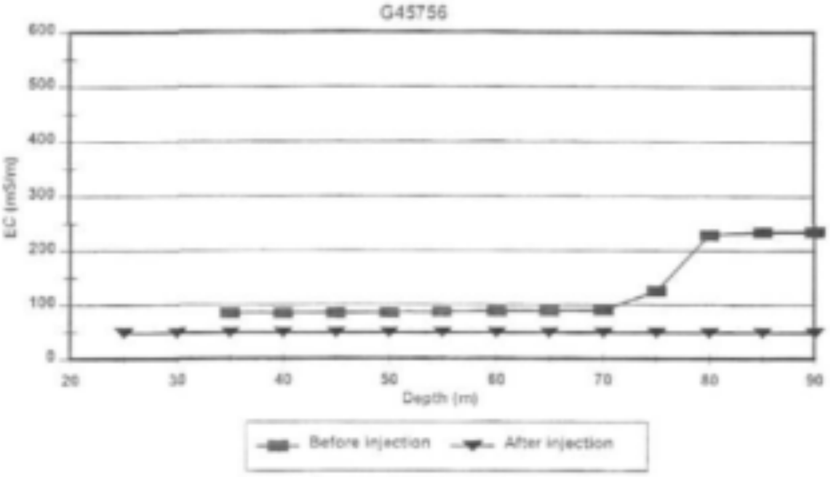

Figure 18 G45756: Electrical conductivity profile before and after injection in 2000

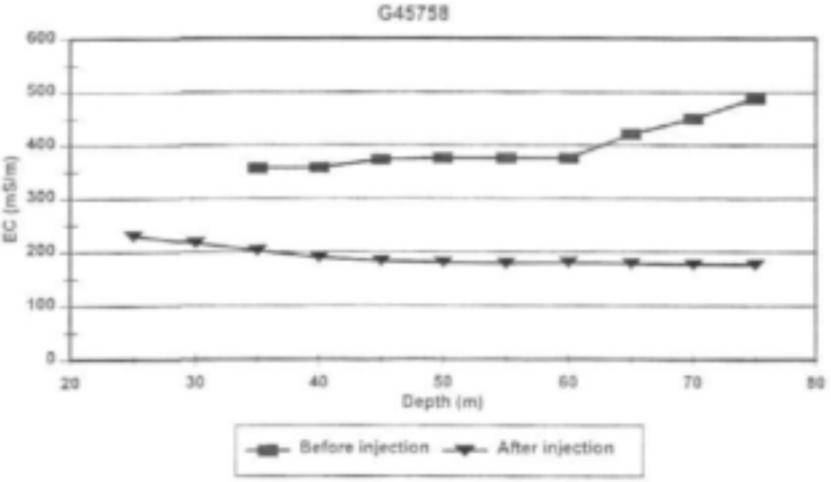

Figure 19 G45758: Electrical conductivity profile before and after injection in 2000

7. GROUNDWATER FLOW CHARACTERISTICS AND HYDRAULIC PARAMETERS

Aquifer parameter values were determined from the injection tests using observation boreholes. Figure 20 presents the results of the 1999 injection test. At first the flow through the sand filter was very weak, but with time it improved and gave a maximum flow of 1.5 L/s, or 130 m^3/day. This was the maximum injection rate the borehole received for this test.

The 1999 test shows that for the first 20-30 minutes, the injected water occupied fracture storage. The T-value is reasonably high (about 100 m^2/day), and the S-value low (about 0.0006). The fracture network appears to be limited in extent, and after this flow period, the build up in pressure forced water into the smaller fractures adjacent to the fault zone. This has a much lower T-value (about 18 m^2/day) and a higher S-value (about 0.001).

The data was not collected for a long enough period to establish the country-rock aquifer parameter values, however, these were expected to be much lower. This was confirmed by the year 2000 injection test where the T-value obtained prior to boundary effects was found to be 1.4 m^2/day (Figure 20). The S-value obtained was 0.002.

The aquifer parameter values are estimated to be:

	Transmissivity m^2/day	*Storativity*
Fracture	~ 100	~ 0.0006
Matrix (or fractured country-rock)	~ 1	~ 0.002

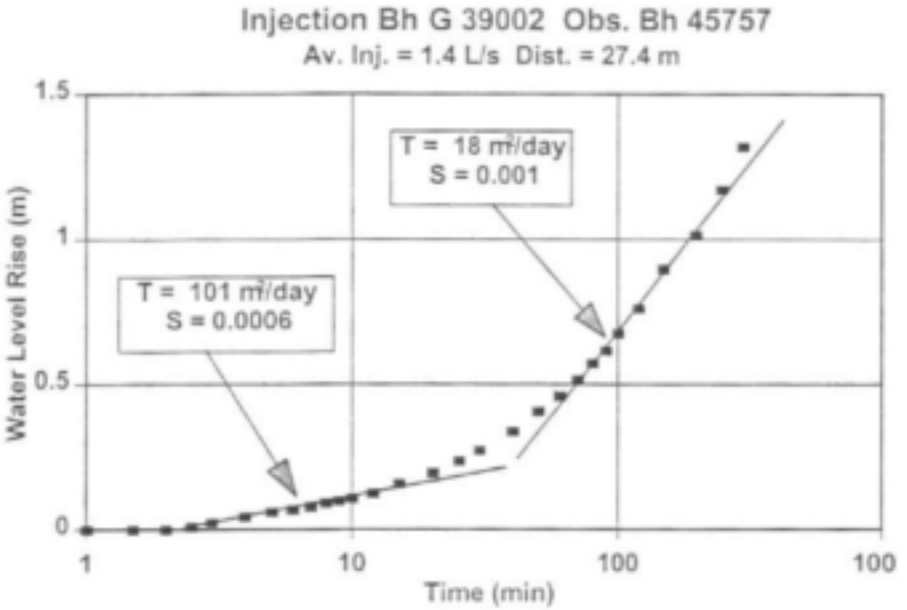

Figure 20 Short-term response in borehole G45757 to injection into G39002

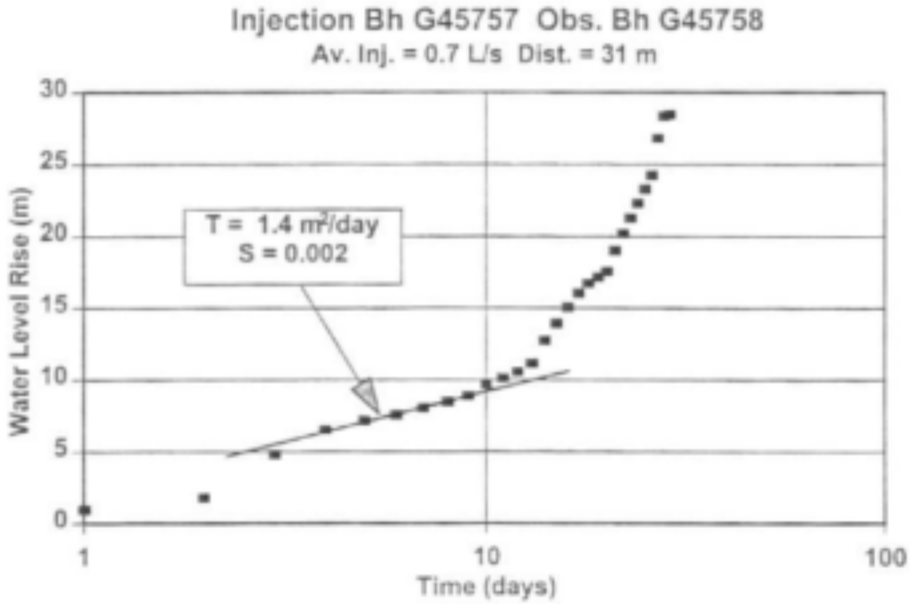

Figure 21 Long-term response in borehole G45758 to injection into borehole G45757

During the 2000 injection test the flow through the filer ranged from 0.5 to 0.7 L/s. After 13 days of injection, it appeared as if closed boundary conditions prevailed. This is evident from the sudden rise in water levels after about two weeks of injection (on the semi-log plot), and the log-log slope which has a value of one during the latter part of this test. The storage "reservoir" is thus believed to be of a limited size, possibly only associated with the fault's fracture zone and the adjacent fractured country rock. If this is indeed the case, it means that the aquifer is unlikely to be able to receive water at a relatively high rate (0.7 L/s) for an extended period of time. This is not much of a concern, since the aquifer accepted the water for 90 days during the year 2000, and it is not that often that the river flows for much longer than three months in any given year.

The notion that the aquifer is closed or very poorly linked to other groundwater bodies is supported by borehole G39002's low sustainable yield. This yield is believed, from historical abstraction and water level data, to be in the order of 0.1 L/s (or about 6.5 m^3/day).

Should long-term injection (for example longer than six months) be possible at some stage in the future, it is quite likely that after the macro- and micro-fractures associated with the fault zone have been filled with water (at a rate of about 0.5 - 0.7 L/s), the rate at which the aquifer will be able to receive water will be in the order of 0.1 L/s.

To conclude this section, the aquifer consists of a permeable, linear fracture zone which is surrounded by relatively impermeable granitic country rocks. The storage capacity of the fractured "reservoir" appears to be limited in extent, but can be filled at a rate of about 0.7 L/s (and possibly slightly higher) by borehole injection. Because the aquifer is vulnerable to over abstraction, artificial recharge can play a major role in maintaining a reasonable yield from this limited groundwater body.

8. CONCLUSIONS AND RECOMMENDATIONS

This test demonstrated that filtered river water can be injected into boreholes at a rate of 0.6 L/s over a period of at least 138 days. This is the average rate at which water enters the boreholes from the sand filter during weeks of injection. The highest injection rate achieved over a shorted period was 1.4 L/s into production borehole G39002. As the efficiency of the sand filter decreased, so the injection rate decreased. In order to maximise the injected volume, it is essential that the efficiency of the sand filter is maintained as described in the operating manual.

Over the past five years (1995 to 2000), the ephemeral river that supplies this borehole flowed for one to six months each year, except in 1998 when there was no flow. Assuming an average injection rate of 0.6 L/s, during a poor rainfall year, when only one month of water is available for injection, 1 550 m^3 can be injected; and during a good rainfall year, when six months of water is available for injection, 9 300 m^3 could feasibly be injected (although the aquifer would need to have been heavily pumped prior to this for the necessary space to be available).

Considering that the borehole's sustainable yield is about 2 400 m^3/a, injection over a one to six month period could account for 0.6 to 3.9 times this. During the 2001 injection run which lasted 138 days, 6 567 m^3 was injected and 1661 m^3 was pumped during the same period (due to the town's reliance on the borehole). The 4 906 m^3 (net) that was injected amounts to twice the borehole's sustainable yield.

An additional benefit of introducing this fresh water to the aquifer, is that it improves the groundwater quality. The test results demonstrated a significant improvement in the quality of the groundwater and the abstracted water. The more water that is injected the more the electrical conductivity will decrease and it will remain at lower levels for a longer period. With repeated injections, it may eventually be possible to maintain a lower electrical conductivity all year round.

This scheme can continue to provide relatively good quality water to Karkams that would otherwise be lost to evaporation. It does however require very basic maintenance before and during injection runs. If this maintenance is not done (as indicated in the operation manual) the sand filter will lose efficiency and the injection rate will be dramatically reduced.

Although the cost of developing this scheme was not determined during this study, this technology is considerably less than the cost of treating or developing other sources. Prior to considering new sources for Karkams or other rural villages, an economic assessment should be carried out (in addition to water resource assessment studies) to establish whether artificial recharge is a cost effective alternative to conventional water supply options.

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APPENDIX 3.1:

Modifications to river filter for augmentation of groundwater at Karkams, Northern Cape

MODIFICATIONS TO RIVER FILTER

for

AUGMENTATION OF GROUND WATER

at

KARKHAMS, NORTHERN CAPE

by

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9 November 1998

MODIFICATIONS TO RIVER FILTER FOR AUGMENTATION OF GROUND WATER AT KARKHAMS, NORTHERN CAPE.

INTRODUCTION

The small town of Karkhams in the Northern Cape depends almost entirely on ground water abstracted from boreholes for its water supply. One of these boreholes (   ) ,which is located very close to a small seasonal river in a valley approximately 15 km east of Karkhams, has its ground water supply augmented by filtered river water which is gravitated into the borehole. The filter, which is located in the actual river bed, is a very basic shallow type sand filter without proper underdrains. The filter suffers damage whenever the river comes down in flood and generally produces a poorly filtered water which often contains fine sand. Another defect in the system is that the filtered water is introduced at the top of the borehole via a hole in the side of the lining. The water pouring into the borehole becomes aerated, a condition which may cause precipitation of iron etc. in the borehole.

An on-site investigation was carried out on 30 October 1998 to see what modifications should be implemented to improve the operation of the system. The existing system, as well as the proposed modifications, are described in this report.

EXISTING SYSTEM

A drawing of the filter/collector is shown in Figure 1.

The existing filter/collector is a cement brick structure built on an irregular rock face in the river bed. This rock face slopes downward in the direction of flow at approximately 7 to 15 degrees. The main body of the unit is on the 15° slope. Only a narrow section of about 0.3 m is on a level portion of the rock adjacent to the front (downstream) wall of the filter unit. The total possible filtration area is 2.75 m by 1.5 m, or 4.125 m^2, but owing to the severe slope, the effective filtration area is only about 1.0 m by 1.5 m, or 1.5 m^2.

The average filter bed depth over the effective part of the filter is only 1.0 m, which limits its efficiency, and assuming that the bed consisted of properly graded sand, would deliver at best approximately 0.45 m^3/h, or 0.125 L/s.

Judging by the condition of the filter when it was visited on 30 October 1998, when the river was completely dry, it probably produced considerably more water than the calculated value, but the filtered water was most likely of poor quality as the bed was partly filled by large boulders and proper underdrains had apparently not been provided. In the past, problems had been encountered when sand had been introduced into the borehole which ruined the submersible pump. This was not surprising, considering the condition of the filter.

The main structure, which comprises cement bricks and mortar built directly on the sloping rock face, shows signs of aggressive attack by the water, which is probably acidic which is typical of surface waters originating in granitic areas. The front wall of the filter had a substantial hole through it and its downstream face had thick white deposits on its surface, which is typical when aggressive water leaches lime from the cement matrix.

The 50 mm galvanized iron pipe leading from the filter unit to the borehole casing was encased in a greased cloth wrapping and had barbed wire wrapped around most of its length. These measures were apparently taken to prevent corrosion and tampering by the nearby inhabitants.

A serious problem with this filter unit is that during a flood situation the water completely swamps the unit, disrupting the filter bed and depositing large boulders all over its surface. It is located directly in the flow path of the water so as to make optimum use of the low flow condition. No provision has been made to divert the flood waters around the unit.

PROPOSED MODIFICATIONS TO THE FILTER

Figures 2, 3 ,4, 5 and 6 show the proposed modifications to the filter unit.

The following modifications and repairs have to be carried out:

(1) The front wall of the filter has to either be completely rebuilt or be effectively repaired. Any damage to the side walls and the widened inlet section walls will also have to be repaired.

(2) A 50 mm Class 9 PVC pipe section must be installed through the front wall such that there is 50 mm clearance between the lower edge of the pipe and the short level section of rock on the inside of the filter (see Figure 2).

(3) The entire inside of the filter must be coated with an epoxy tar and suitable lining material, used to waterproof water storage tanks and roofs, including the base rock area. An epoxy tar that is suitable for tanks that are used for drinking water must be used.

(4) A slotted 50 mm PVC manifold is to be constructed as per Figure 3 and attached the pipe fitted through the wall. The manifold is to be supported off the rock base by suitable supports (these could be cast from mortar or made up from PVC material).

(5) The bottom section of the filter is to be filled with fairly coarse gravel (5 to 10 mm diameter) to a depth of 50 mm above the top edge of the slotted PVC manifold. This coarse fill must be put in carefully so as not to damage or disturb the slotted manifold. The fill layers are shown in Figure 4.

(6) On top of the coarse gravel a layer coarse sand (1 to 2 mm) must be filled in to a depth of 100 mm (above the gravel). The gravel and the coarse sand make up the support bed of the filter (see Figure 4).

(7) The rest of the filter must then be filled in with graded sand (0.2 to 0.4 mm) such that there is about 25 mm freeboard above the bed, i.e. clearance between the left (lower) wall top edge and the level surface of the sand. This is the actual filter bed and must be maintained in a level condition.

(8) The outlet pipe (50 mm PVC) must then be adapted to the existing metal pipe, preferably by means of a PVC union so that it can be easily uncoupled if, at a later stage, the metal pipe has to be replaced. Alternatively, the union can in turn be adapted to a HDPE pipe to

replace the metal pipe. The HDPE pipe can then be led, via a suitable elbow at the top of the borehole casing, to below the water level in the borehole.

(9) The pipe conducting the tail water to the borehole must incorporate a shut off valve, an adjustable control valve (a brass gate valve will be acceptable) and a totalizing flow meter (typically a Kent domestic water meter). It would be advisable to have these fittings near to the pump house. A drain valve should also be fitted (see Figure 5).

(10) To prevent the storm flow from washing right over the filter unit, a "wall" of gabions must be arranged at an angle across the inlet area of the filter, as shown in Figure 6. The gabions may be assembled on site, using suitable strong wire mesh to enclose stones of approximately 50 to 150 mm diameter. The outer layer of gabions which will be hit directly by the flood water would preferably contain larger stones, say 100 to 150 mm, while the gabions along the inner layer should contain smaller stones of 50 to 100 mm diameter. To prevent the gabion wall from being broken up and washed away, the components should be tied together with stout wire or even steel cables. These wires or cables should be anchored around fixed objects (either suitable rocks, or anchor points inserted into the rock) to prevent the wall from being swept away when the water level is high.

(11) During flood situations, a lot of debris will be swept against the wall, or some may even be swept over it onto the filter bed. As soon as the main flood water subsides, it will therefore be essential to remove the debris from the filter surface and from the gabion wall, so that the normal flow is not diverted and can pass onto the filter. If necessary, the top of the filter bed must be levelled off or topped up to its previous level to ensure proper filtration.

GENERAL MAINTENANCE

The filter unit must be maintained properly if it is expected to produce water of a suitable quality for augmenting the ground water. It is a very shallow filter having very little effective filtration area, so it is essential that its surface is kept clean and free of debris and stones. The upper layer of fine sand will become clogged after some time and will have to be carefully scraped off to a depth of about 50 mm and clean graded sand added to restore the surface to its former level of 25 mm below the left (lower) sidewall.

If the tail water flow from the filter decreases, the surface sand is probably clogged and has to be renewed as described above.

If any leakage is noted around the filter base or through the filter walls, it will be necessary to repair this from the inside. It will therefore be required to remove the filter bed and the support bed, repair the leak and reinstate the bed.

The gabion wall must be inspected frequently and repaired where necessary. If it is allowed to deteriorate it will be washed away and the filter will be damaged.

Samples of the filtered water should be taken and tested to ensure that the filter is functioning effectively. If properly maintained, it should produce clear, silt free water at a rate of approximately 0.25 to 0.5 L/s.

NOTES

A number of options was considered before it was decided to upgrade the existing filter unit as described above. Options such as enlarging the filtration area by rebuilding the filter unit, collecting the river water higher up in the catchment area to provide sufficient head to operate a pressure sand filter located next to the borehole, diverting a portion of the river flow away from the present water course and passing it over a properly constructed crossflow filter, were all considered. The cost of these options would all be considerably more than upgrading the existing filter.

The upgraded unit as described in this report will, however, have limitations.
Some of these are:

- The quality of the tail water produced will not always be of a high standard. When the water level in river is high, it will most likely be very turbid. The filter will remove most suspended material, as well as fine sand which in the past has damaged the pump. It is unlikely that it will remove colloidal matter, such as fine clay particles generally associated with turbidity. Colloids have to be destabilised by addition of coagulants such as aluminium or iron salts and coagulated to form larger particles before they can be removed by sand filtration.

- The filter is not a simple depth filter due to its tapering section as viewed from the side. The filtering action will most likely be a combination of depth and longitudinal filtration. This will have the effect that the output may be higher than expected, but the effect on the water quality is unknown.

- The weak point in the design is the durability of the gabion wall under storm flow conditions. Ii is important that this item must be well constructed and if possible, firmly anchored. If it washes away, the filter bed will be totally disrupted and, at worst, the filter structure damaged.

- The success of this filter/collector system depends on how well it is maintained. The gabion wall must be regularly inspected and maintained. The filter must be kept clean, with the top of the sand bed level and the correct distance below the overflow level. The top of the bed must be scraped off and replaced with similar graded sand when the filter output drops below an acceptable level.

- The pipes and valves must be kept in good condition. If air gets trapped in the pipe from the filter or in the pipe down the borehole, it will restrict or completely stop the water flow. Air must be bled out when this occurs.

H A de Villiers

9 November 1998

APPENDIX 3.2:

Geological logs of boreholes G45755, G45756, G45757, G45758

Well Log: Lithology & Construction

Well Ident G45755	Name **Karkams**

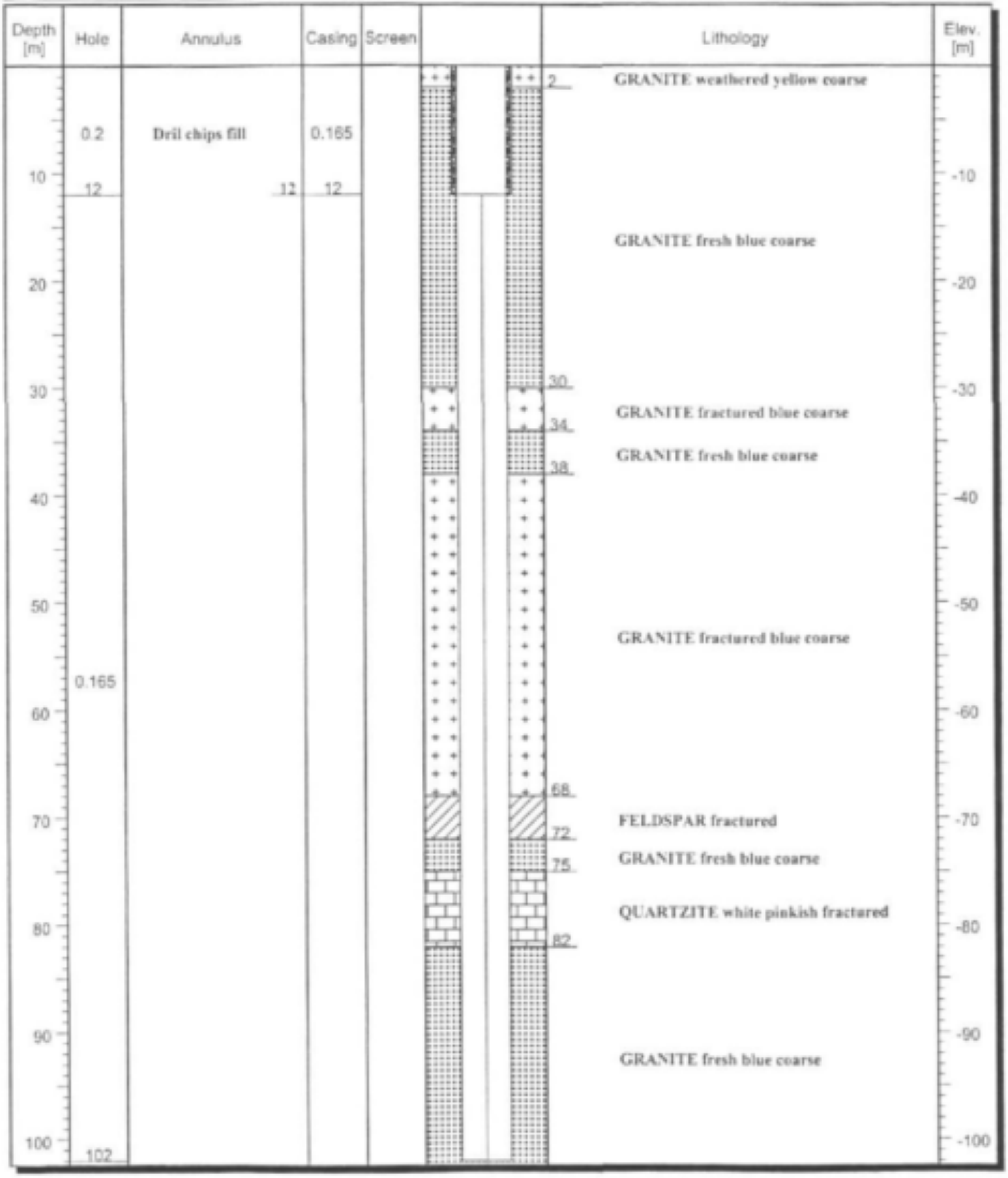
Drill. Method	**Air percussion**	Drill. Dates	**Aug 1999**

X	Y	Z	Meas. Pt. Elev.

All measurements are in meters. Hole and casing diameters in inches.

Scales (1: xxx)

Water Level (m AMSL)	Vertical	Horizontal



Well Log: Lithology & Construction

Well Ident G45756

Name: Karkams

Drill. Method: **Air Percussion**

Drill. Dates: **August 1999**

X

Y

Z

Meas. Pt. Elev.

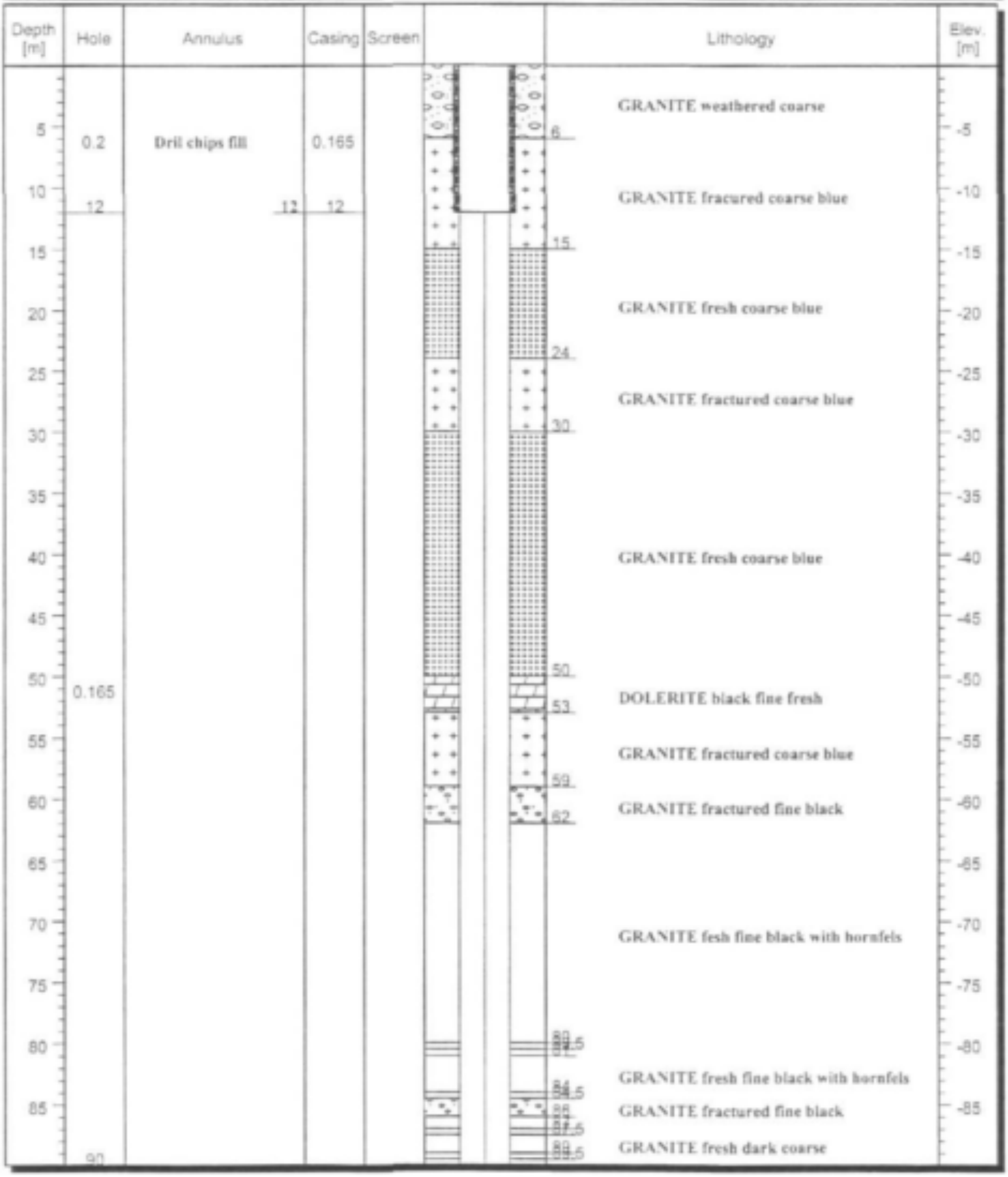
All measurements are in meters. Hole and casing diameters in inches.

Scales (1: xxx)

Water Level (m AMSL)

Vertical

Horizontal



Well Log: Lithology & Construction

Well Ident G45757

Name Karkams

Drill. Method Air percussion

Drill. Dates August 1999

X

Y

Z

Meas. Pt. Elev.

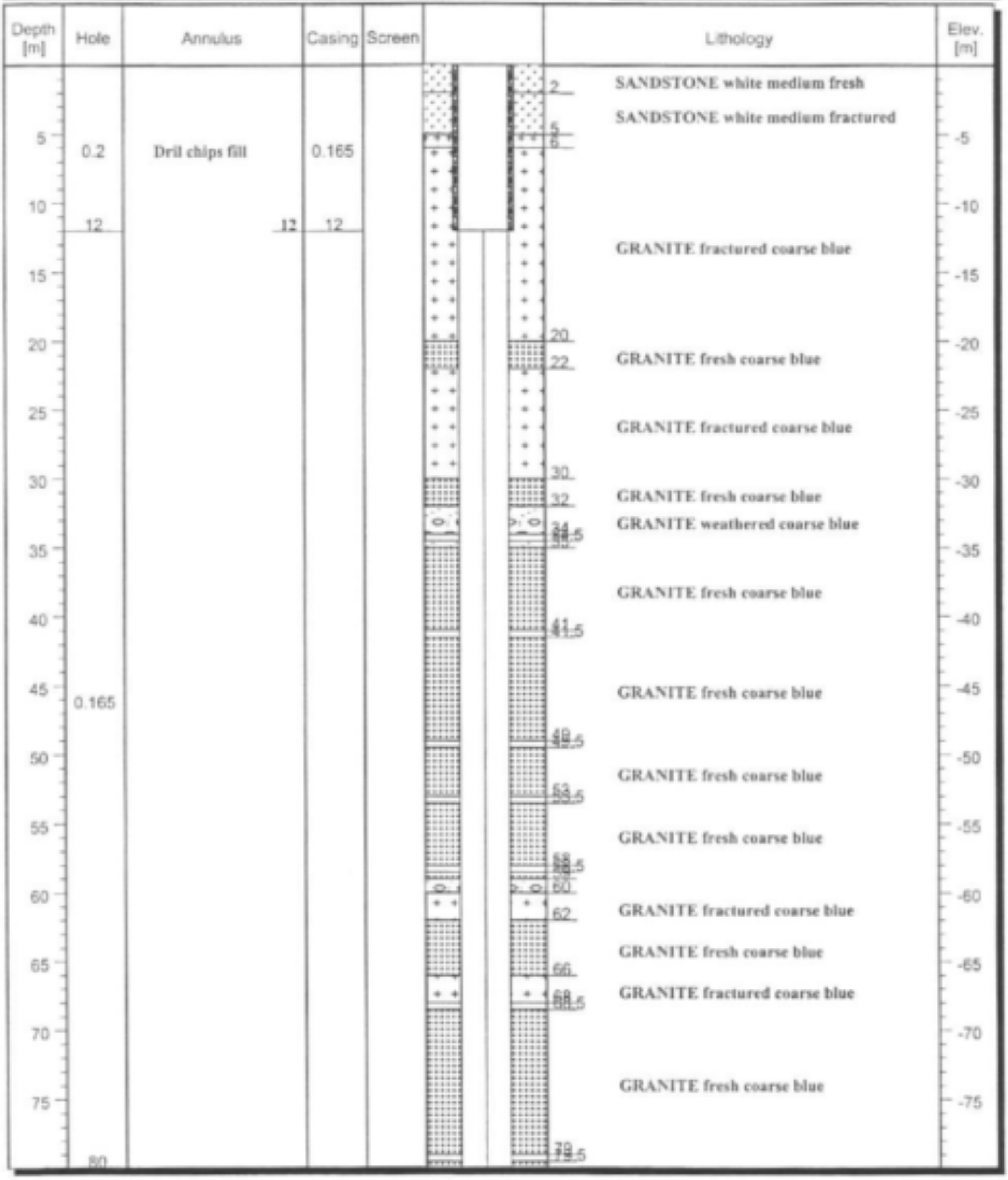
All measurements are in meters. Hole and casing diameters in inches.

Scales (1: xxx)

Water Level (m AMSL)

Vertical

Horizontal



SANDSTONE white medium fresh

SANDSTONE white medium fractured

GRANITE fractured coarse blue

GRANITE fresh coarse blue

GRANITE fractured coarse blue

GRANITE fresh coarse blue

GRANITE weathered coarse blue

GRANITE fresh coarse blue

GRANITE fresh coarse blue

GRANITE fresh coarse blue

GRANITE fresh coarse blue

GRANITE fractured coarse blue

GRANITE fresh coarse blue

GRANITE fractured coarse blue

GRANITE fresh coarse blue

Well Log: Lithology & Construction

Well Ident G45758	Name **Karkams**
Drill. Method **Air percussion**	Drill. Dates **August 1999**

X	Y	Z	Meas. Pt. Elev.

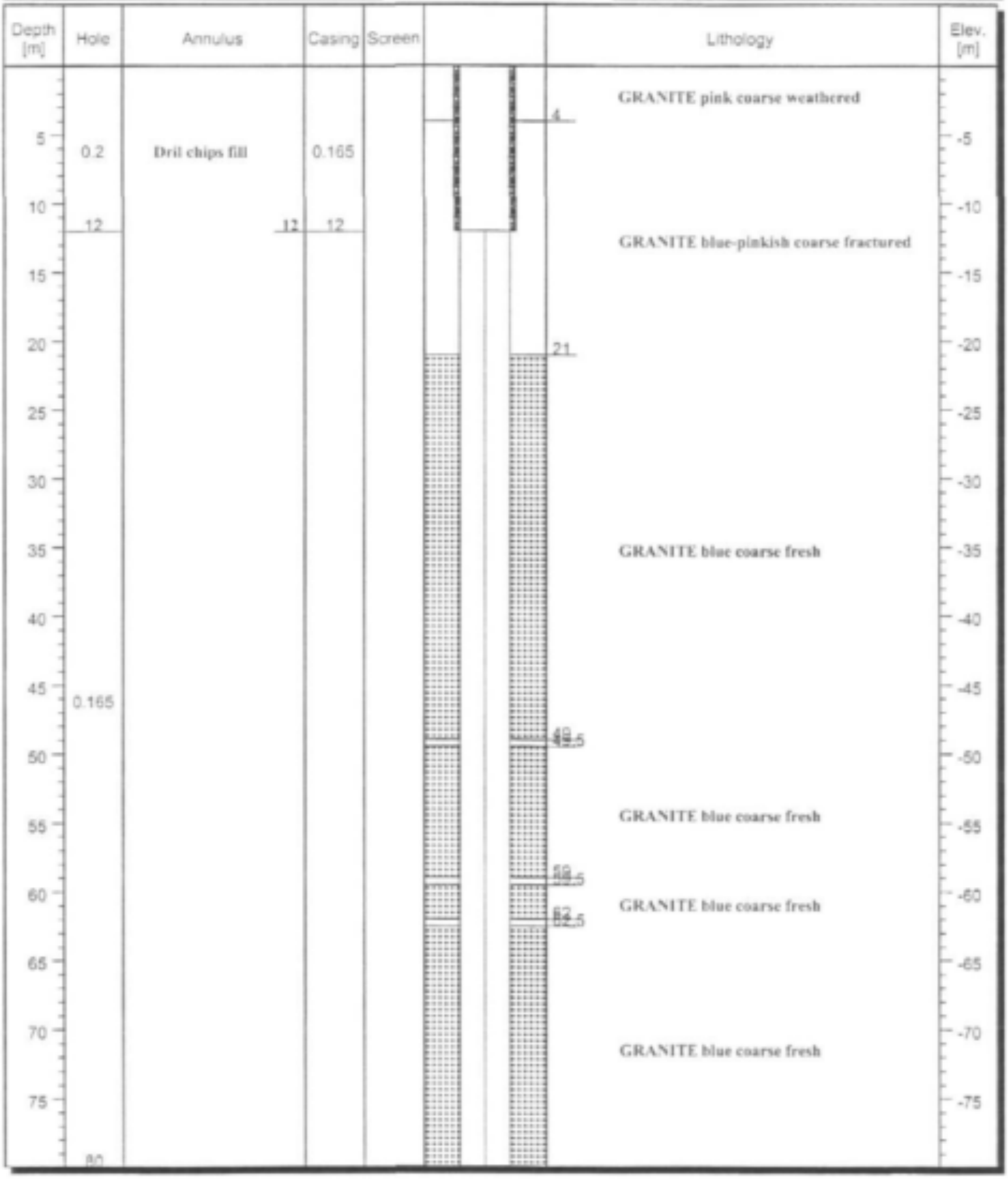
All measurements are in meters. Hole and casing diameters in inches.

Water Level (m AMSL)

Scales (1: xxx)

Vertical

Horizontal



APPENDIX 3.3:

Geophysical log of boreholes G45755, G45756, G45757 & G45758

DEPARTMENT OF WATER AFFAIRS AND FORESTRY

GEOHYDROLOGY - BOREHOLE TECHNOLOGY

Province: NORTH CAPE   District: GARIES   Farm: KHARKAMS

Borehole Number: G45755   Owner Name: -

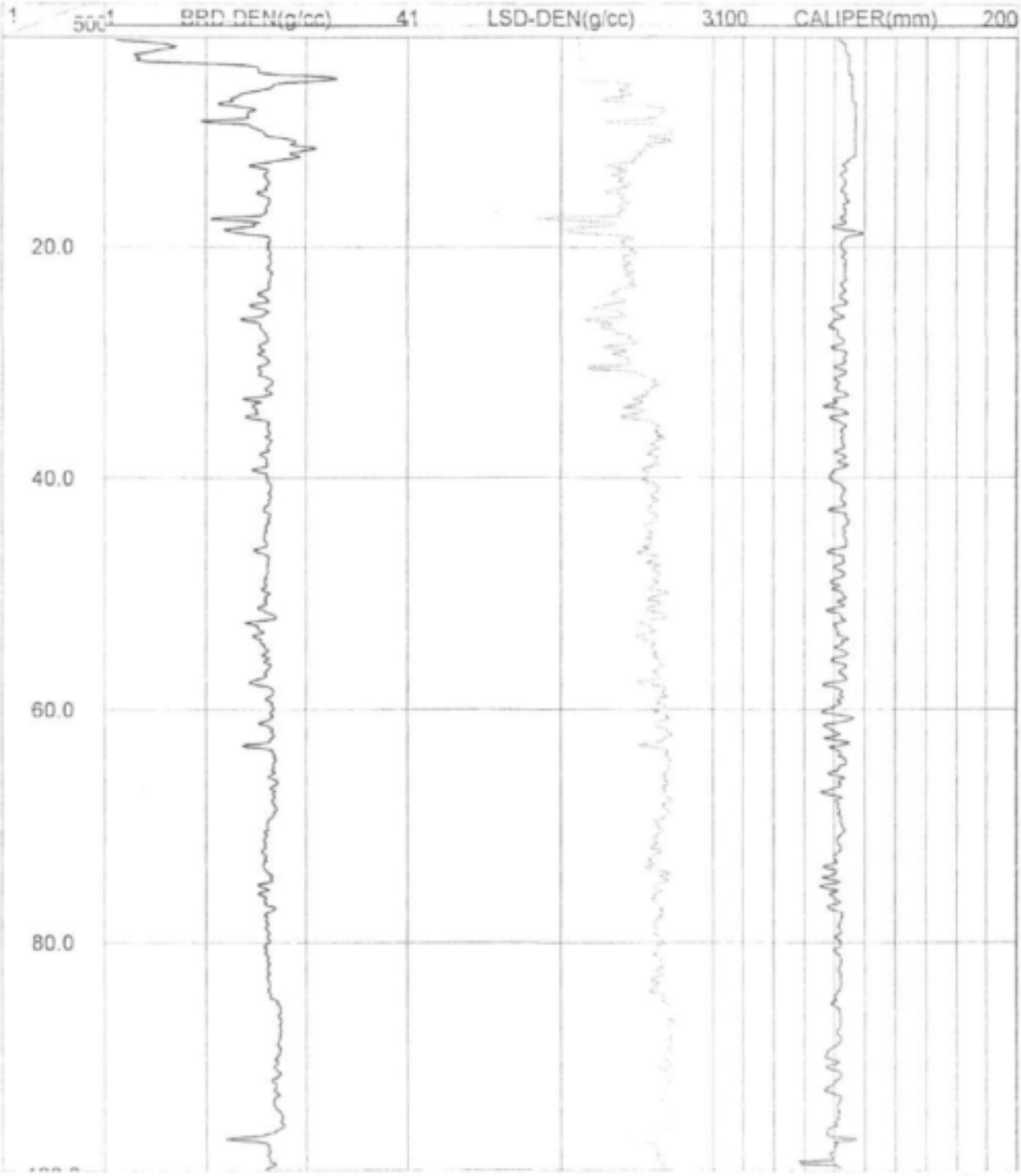
Date: 12.01.2000   Co-ordinates: S 30 19 41.0   E 18 01 14.6

Geophysical Log by: BLV   Log Serial Number: 2784

Main Formation Drilled: - [illegible]

Depth Drilled (m): - 107 m   Borehole Yield (l/s): - 0

Casing Installed (m): - 12 m   Water Level (m): -



DEPARTMENT OF WATER AFFAIRS AND FORESTRY

GEOHYDROLOGY - BOREHOLE TECHNOLOGY

Province: [illegible] District: GARIES Farm: [illegible]

Borehole Number: [illegible] Owner Name: -

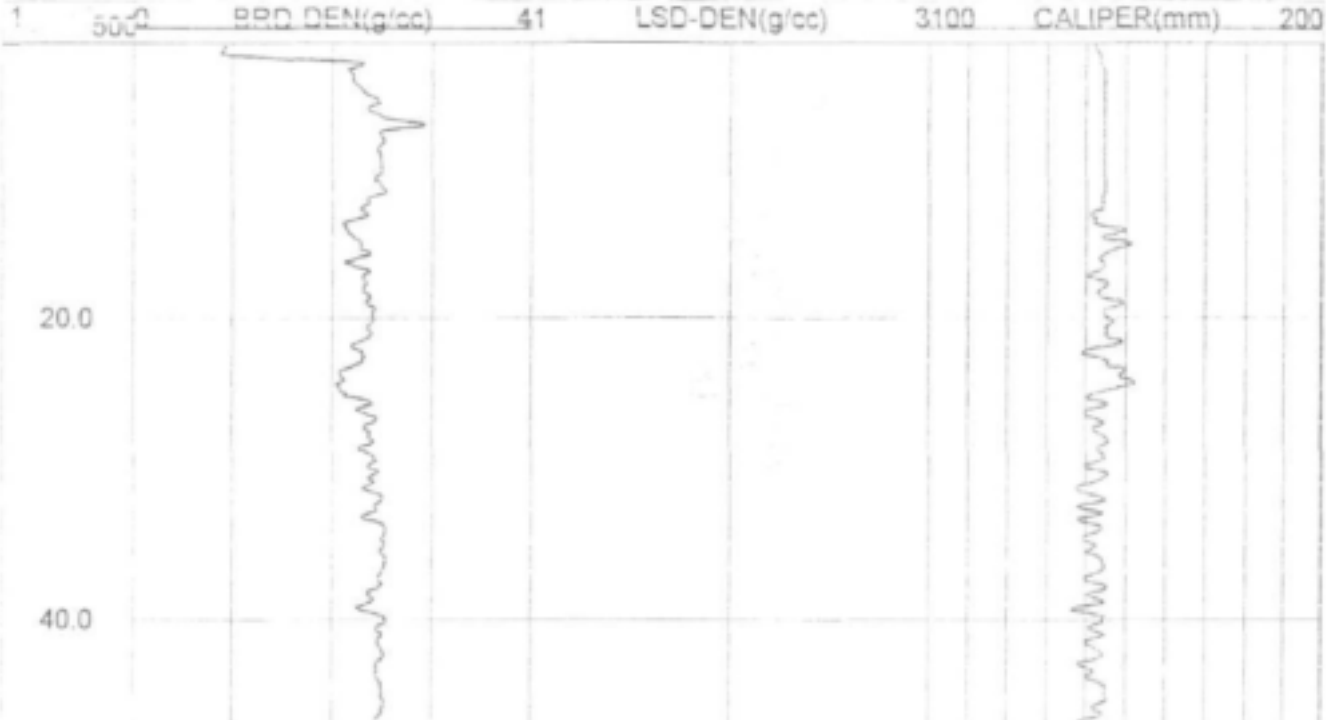
Date: [illegible] Co-ordinates: [illegible]

Geophysical Log by: [illegible] Log Serial Number: [illegible]

Main Formation Drilled: [illegible]

Depth Drilled (m): [illegible] Borehole Yield l/s: [illegible]

Casing Installed (m): [illegible] Water Level (m): [illegible]



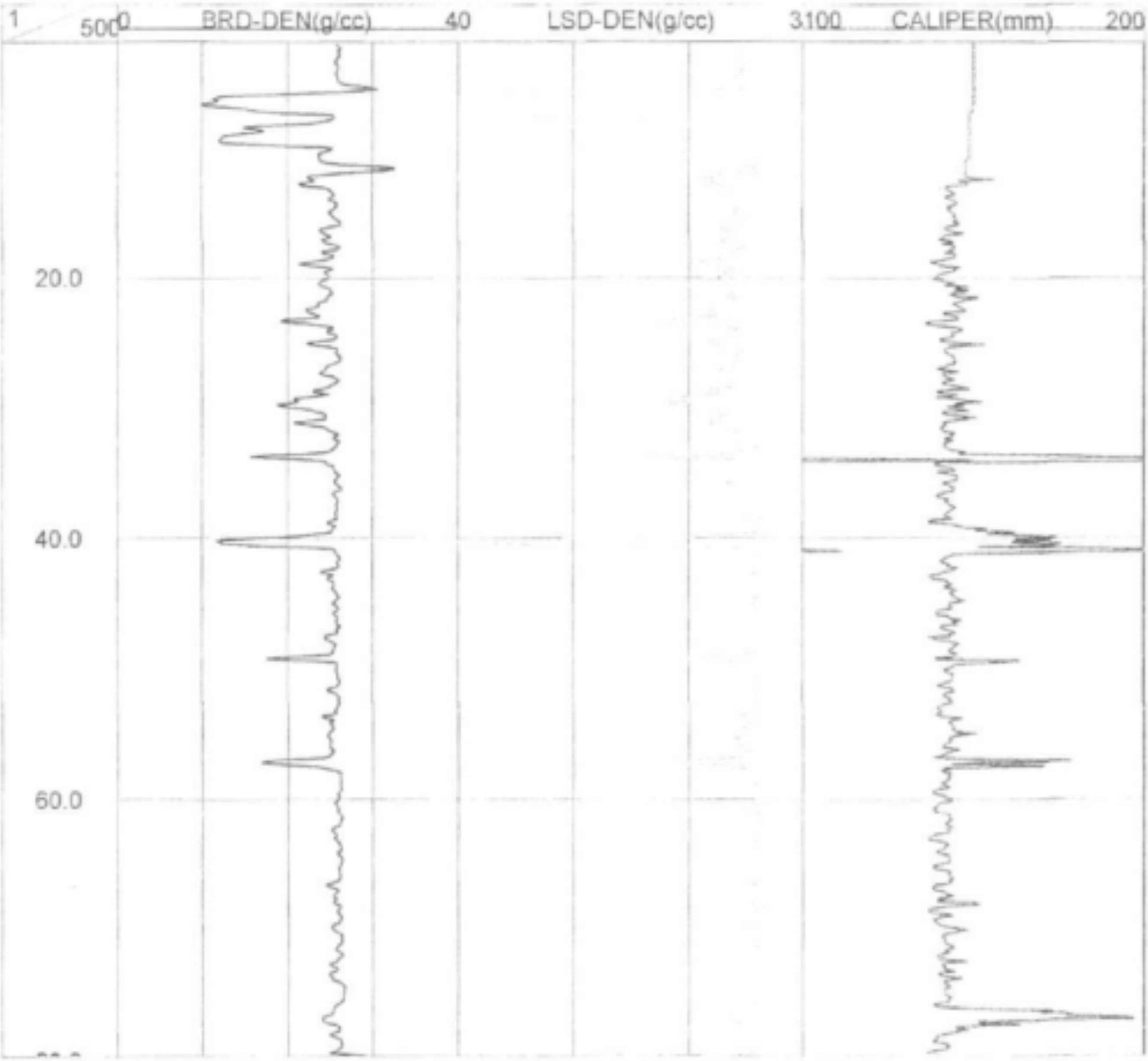
DEPARTMENT OF WATER AFFAIRS AND FORESTRY

GEOHYDROLOGY - BOREHOLE TECHNOLOGY

Province : NORTH CAPE
District : GARIES
Farm : KHARKAMS

Borehole Number : G45757
Date : 12.01.2000
Geophysical Log by : BLV
Owner Name : -
Co-ordinates : S 30 19 40.8 E 18 01 14.6
Log Serial Number : 2788

Main Formation Drilled : -
Depth Drilled (m) : -
Casing Installed (m) : -
Borehole Yield (l/s) : -
Water Level (m) : -



DEPARTMENT OF WATER AFFAIRS AND FORESTRY

GEOHYDROLOGY - BOREHOLE TECHNOLOGY

Province: NORTH CAPE District: GARIES Farm: KHARKAMS

Borehole Number: G45706 Owner Name: -

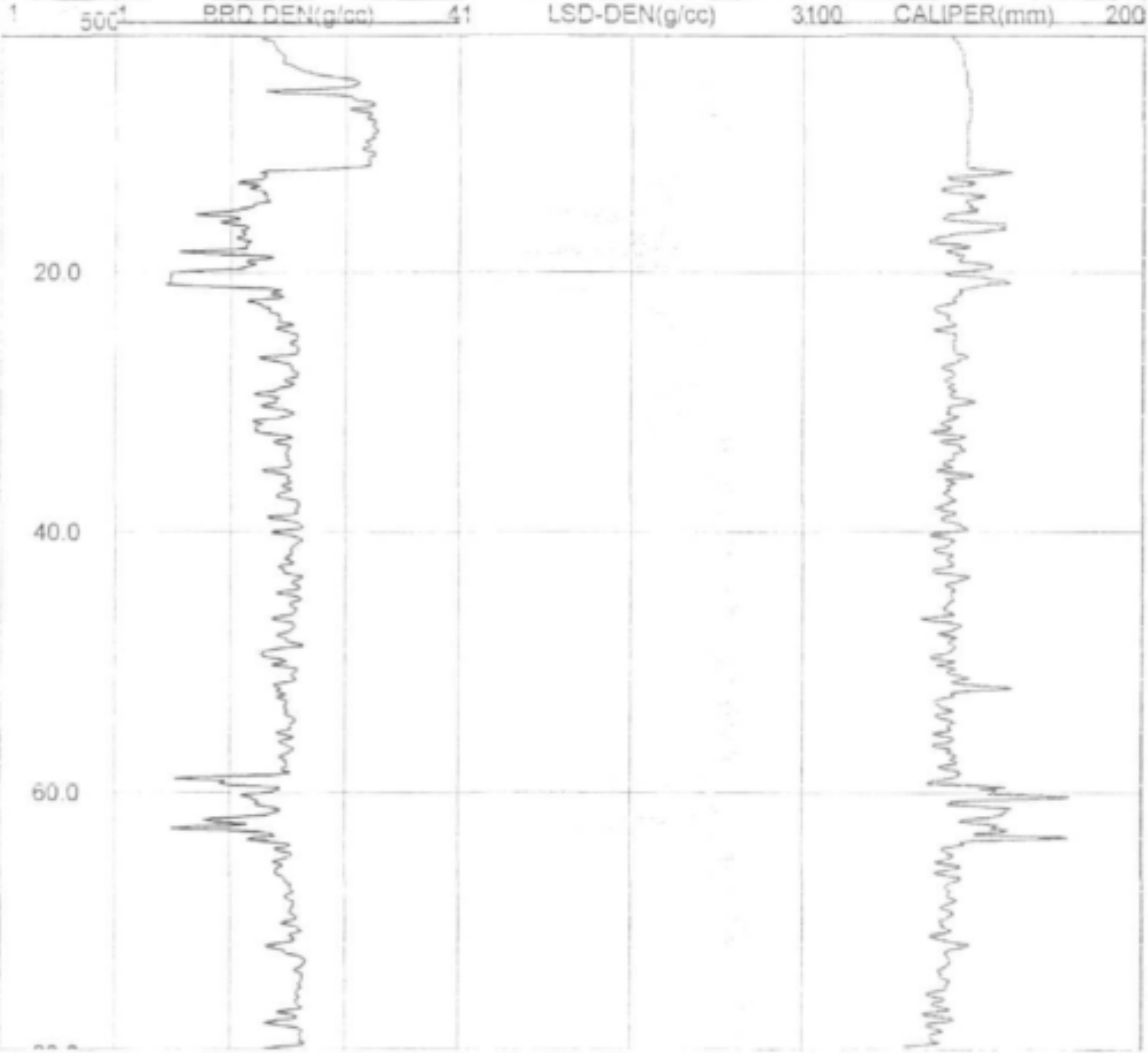
Date: 12.01.2000 Coordinates: S 30 19 40.7 E 18 01 14.5

Geophysical Log by: BJV Log Serial Number: 2797

Main Formation Drilled: [illegible]

Depth Drilled (m): [illegible] Borehole Yield (l/s): [illegible]

Casing Installed (m): [illegible] Water Level (m): -



APPENDIX 3.4:

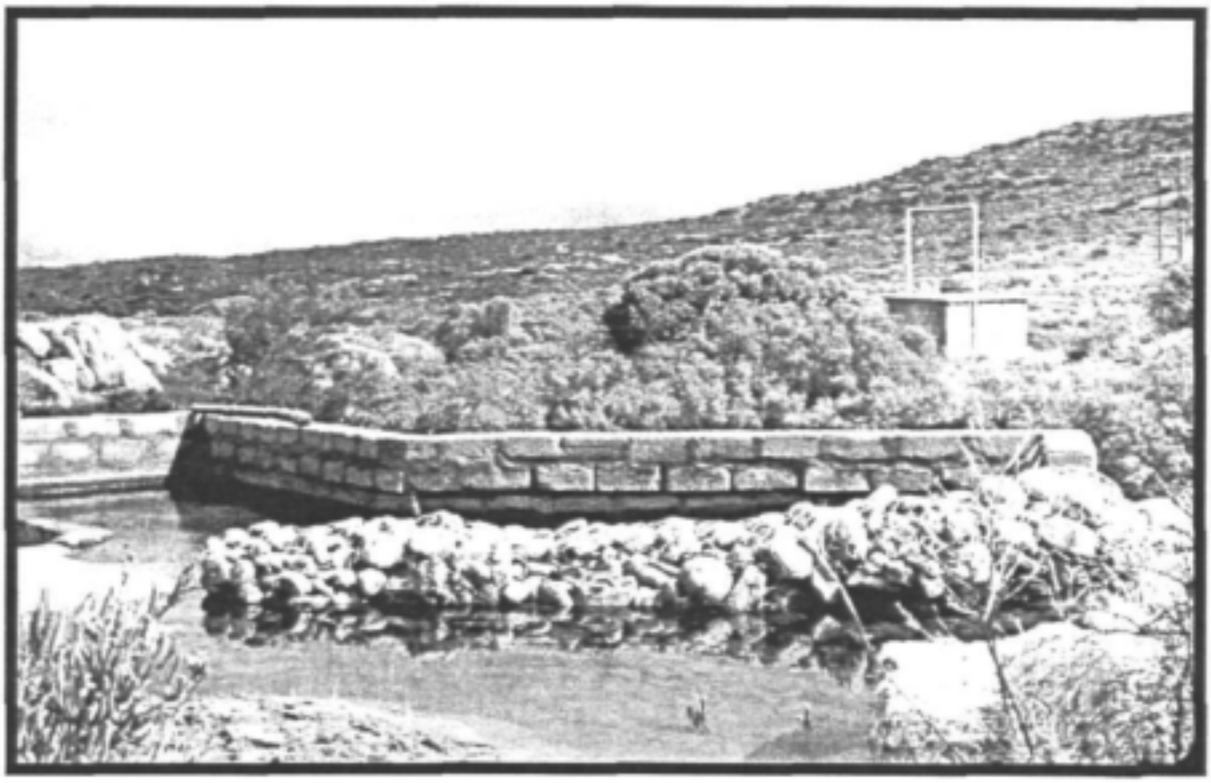
Manual on artificial groundwater recharge in Karkams (in Afrikaans)

Handleiding vir Kunsmatige Grondwateraanvulling in Karkams

deur

Ricky Murray & Pannie Engelbrecht
WNNR, Stellenbosch

Junie 2001



Die sandfilter waardeur die water filtreer voordat dit by boorgat G45757 of G39002 aangevul word. In die agtergrond is die pomphuis van produksieboorgat G39002 sigbaar

Die skema

'n Skematiese voorstelling van die Karkams kunsmatige grondwateraanvullings- skema verskyn hier onder.

Die sandfilter
Die aanvullingsboorgate:
  - G45757 (geboor 1999)
  - G39002 (bestaande onttrekkingsboorgat)

Die onttrekkingsboorgat:
  - G39002

Die waarnemingsboorgate:
  - G45755
  - G45756
  - G45758

Uitleg van die kunsmatige grondwateraanvullingsskema.

Nota: Woordelys (Tegniese terme), met fotos uitgebeeld, verskyn agter in die handleiding.

Heel jaar take

Watervlak, onttrekking, water gehalte, en reënvalmonitering

Hierdie take moet gedurende die hele jaar gedoen word, al vind daar geen kunsmatige grondwateraanvulling plaas nie.

Die watervlakke in al die boorgate rondom die skema reageer basies dieselfde gedurende onttrekking of aanvulling, dus hoef net een boorgat naamlik G45756 se watervlak gemoniteer word.

Wat om te meet.	**Wanneer**	**Opmerkings**
Watervlak in G45756	Weekliks	Maak 'n nota van die fase waarin die boorgat was het toe die lesing geneem is (onttrekking, rus of aanvulling). Watervlakke moet vanaf die bekhoogte van die staalvoering geneem word
Onttrekkingsvloeimeterlesing	Weekliks	Skryf in notaboek
Een water monster	Maandeliks	Ontleed maandeliks vir Elektriese Geleiding en ontleed elke derde maand vir Fluoried
Reënval	Daagliks	Skryf in notaboek

Nota : 'n Vorm waarin bogenoemde data aangeteken kan word, verskyn agter in die handleiding.

Droë seisoen take

Hierdie take moet gedoen word wanneer die sandfilter totaal droog is.

Skanse

Ondersoek die skanse vir skade of verwering en herstel indien nodig. Alle opdrifsels teen die skanse moet verwyder word.

Inlynfilter

Spoel die siffilter skoon met water en verwyder enige sand uit die houer.

Sandfilter

Verwyder 50 millimeter (mm) van die riviersand wat die bolaag van die filter uitmaak en gooi dit stroomaf in die rivierbedding.

Verwyder nou alle sand en gruis uit die sandfilter. Wees baie versigtig om nie die sandfiltervoering onder in die sandfilter te beskadig nie.

Hou die riviersand, growwe sand en gruis in aparte hopies.

Gaan die sandfiltervoering en sandfilterafvoerpyp versigtig na vir skade en herstel indien nodig.

Kyk of die sandfiltervoering stutte in plek is en dat dit die voering goed ondersteun.

'n Tuinslang kan aan die monsternemingskraan van die produksieboorgat gekoppel word en die water kan dan gebruik word om die sandfilter skoon te spoel indien dit sou nodig wees. Die water moet deur die afvloeikraan gedreineer word.

Vul die area onder die sandfiltervoering met growwe gruis (padgruis) (5 – 10mm) tot 50mm bo die voering. Skraap gelyk met 'n hark of plank.

'n 100mm laag growwe sand (1 to 2mm) moet bo-op die padgruis geplaas en gelyk gemaak word. Indien daar nie genoeg growwe sand oorgebly het om 'n 100mm laag te vorm nie moet addisionele sand aangekoop word by "Consol Industrial Minerals" by telefoon nommer 021 691 0010.

Riviersand moet bo-op die growwe sand geplaas word tot en met 25mm onder die oorloop.

Sif riviersand soos benodig. Gebruik die 2 vierkante millimeter roesvrye sif met 2mm gate en skoon riviersand. Gooi alle materiaal wat kleiner is as 2mm aan die afvloei kant van die sandfilter weg. Gebruik net die sand wat bo-op die sif agterbly.

Vloeirigting en Afvloeikkleppe

Maak alle kleppe toe om te verhoed dat vuil water by die produksie- of kunsmatige aanvullingsboorgat inloop. Dit sal ook verhoed dat insekte of klein diertjies in die betrokke boorgate beland.

Nat seisoen take

Die take moet begin sodra die rivier afkom en volgehou word totdat die kunsmatige grondwateraanvulling gestaak word.

Sodra die rivier afkom

Maak die afvloeikraan oop en laat die sandfilter water wegvloei totdat die water helder is.

Toets of die water geskik is vir aanvulling deur water vanaf die afloopkraan in 'n wit plastiek melkemmer of 'n wit plastiek beker (kan van enige Agrimark of supermark aangekoop word) te gooi. Indien die bodem duidelik sigbaar is en geen fyn sand teenwoordig is nie, is die water geskik vir aanvulling.

Sodra die water helder is doen die volgende

1 Teken die aanvullingsvloeimeter lesing aan in 'n notaboek.
2 Stel die vloeirigtingkleppe sodat die filter se water in die aanvullingsboorgat in loop.
3 Maak boorgat G45757 se luguitlaat kraan oop.
4 Maak die afvloeikraan toe en die vloeikraan oop.
5 Maak seker dat die water in die aanvullingsboorgat invloei. Luister by die bek van die boorgat of watervloei gehoor kan word. Luister ook of 'n suiggeluid by die luguitlaatkraan gehoor kan word.
6 Wanneer die water in die aanvullingsboorgat inloop, maak die luguitlaatkraan toe.
7 Kyk wat die vloeimetertempo is en skryf lesing in die notaboek.

Sandfilter

Fyn sand en modder wat deur die rivier afgespoel word, sal die sandfilter stelselmatig verstop. Dit is 'n normale proses en kan redelik maklik reggestel word.

Hierdie taak is uiters belangrik om die gehalte en vloeitempo van die aanvullingswater te verhoog.

As die vloeimetertempo tot onder 0.5 L/s daal, moet die volgende take verrig word:

1 Maak die afvloeikraan oop en maak die vloeirigtingskleppe toe.
2 Kyk bo-op die sandfilter vir enigiets wat die vloei kan belemmer en verwyder dit (bv. blare, bosse, klippe, gras en sakke ens.)
3 Verwyder alle modder en ander fyn materiaal wat op die river sand sigbaar is. Dit word gedoen deur 'n hout spaander, 'n graaf of 'n stuk LDPE pyp te gebruik om die water wat oor die riviersand se oppervlakte vloei te roer sodat dit die fyn modder van die sandfilter oppervlakte oplig en dit dan oor die oorloop afspoel. Die volgende stap is om die oppervlakte van die rivier sand te versteur sodat dit die onderliggende fyn materiaal los maak. Roer weer die water sodat dit dan oor die oorloop wegvloei. Hierdie process moet herhaal word totdat die rivier water wat deur die sandfilter vloei skoon is. Soos die riviervloei verminder sal die proses 'n bietjie langer neem. Die hele idee is om die water bo-op die sandfilter so troebel as moontlik te maak sodat die modder en ander fyn materiaal kan weg spoel
4 Alge ('n groen slym wat op die riviersand sigbaar sal word) is nog 'n probleem wat

die vloei deur die filter verminder. Die groen slyk moet van die oppervlak van die sandfilter verwyder word met 'n hark of graaf.

5 Enige versteuring van die sandfilter oppervlak sal daartoe lei dat troebel water deur die sandfilter kan vloei. Sodra die skoonmaak proses voltooi is moet die troebelheid van die sandfilter water by die afvloeikraan met 'n wit plasitiek melkemmer of beker getoets word.

6 Wanneer die water helder is, moet die afvloeikraan toe gemaak word en die vloeikraan oop gemaak word.

7 Maak die luguitlaatkraan oop.

8 Maak seker dat die water in die aanvullingsboorgat invloei.

9 Maak die luguitlaatkraan toe.

10 Neem 'n vloeitempo lesing by die vloeimeter en kyk vir 'n verbetering.

11 Herhaal die hele proses indien daar geen verbetering sigbaar is nie.

Die volgende take moet weekliks uitgevoer word terwyl die rivier vloei:

Waarnemingsboorgate

Meet die watervlakke in boorgat G45756 noukeurig en hou rekord van die datum, tyd en diepte onder die bekhoogte in 'n notaboek.

Neem 'n watervlaklesing by een van die ander boorgate elke tweede week om die gelyke styging of daling na te gaan. Die verskil in watervlakke gemeet vanaf G45756 tot die ander boorgate behoort die volgende aan te dui.

G39002 - 1.1m
G45755 - 0.7m
G45757 + 0.3m
G45758 + 0.6m

Filtervloeitempo

Neem gereelde vloeimeterlesings en teken aan op die aangehegde vorm en hou op lêer saam met die watervlaklesings.

Inlynfilter

Spoel die siffilter skoon met water en verwyder enige sand uit die houer.

Skanse

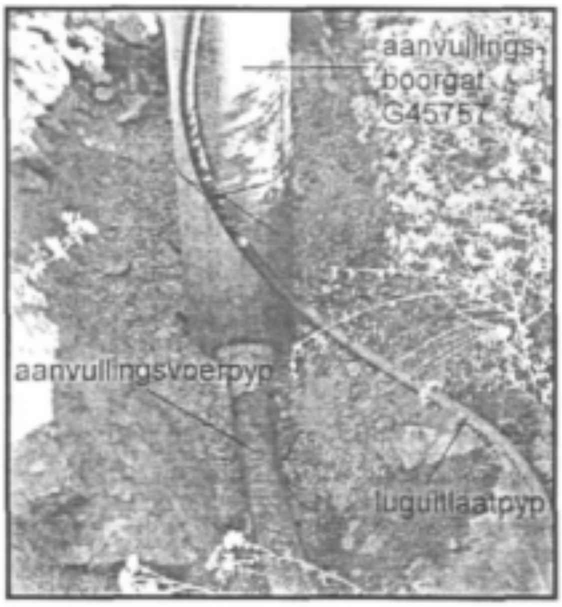
Ondersoek gereeld en verwyder enige opdrifsels wat die vloei na die sandfilter kan belemmer.

Pype

Gaan alle pype se lasse na vir lekkasies en herstel so gou as moontlik.

Karkams Filter woordelys

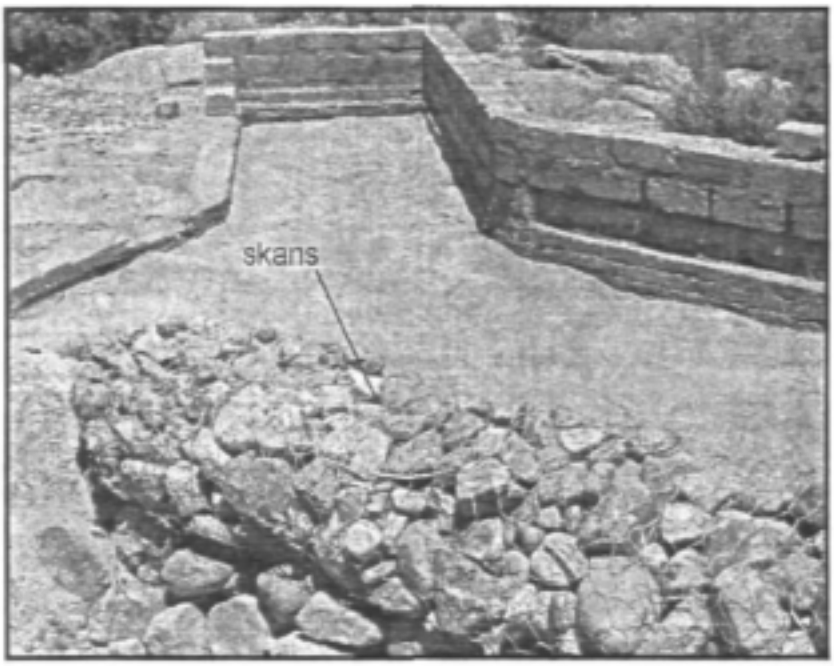
Aanvullingsboorgat:	Boorgat waardeur kunsmatige grondwateraanvulling plaasvind. Gekoppel aan die sandfilter met die aanvullingsvoerpyp en toegerus met 'n luguitlaatkraan.
Aanvullingspyp:	50 meter x 50 millimeter "Layflat tipe" blou pyp. Geïnstalleer binne-in die aanvullingsboorgat om belugting van die water te beheer gedurende aanvulling.
Aanvullingsvoerpyp:	50mm LDPE pyp. Dit lei vanaf die vloeikraan na die aanvulllingsboorgat.



Aanvullingsvloeimeter:	40mm Kent vloeimeter. Geïnstalleer om die filtervloeitempo en volume te meet.

Vloeimeter in die voorgrond met die waarnemings- en aanvullingsboorgate in die agtergrond

Afvloeikraan:	50mm kraan by die eerste T-aansluiting op die filtervloeipyp. Word gebruik om die watervloei na filtrasie van die aanvullingsboorgate af te keer gedurende die filter skoonmaak proses.
Bekhoogte:	Die borand van die staalpyp van 'n boorgat wat bo die grond uitsteek.
Filter:	Sement konstruksie met sifvoering, gevul met sand en gruis.
Filtervloeitempo:	Die tempo word bepaal deur die tyd wat dit neem vir een omwenteling van die tien liter wyser op die vloeimeter in sekondes met 'n stophorlosie te meet.
Sandfiltervoerpyp:	50mm Swart LDPE pyp vanaf sandfilter tot by 1ste T- aansluiting.
Skanse:	Rivier klippe wat met heining draad gebind is. Dit verseker n beskermde, eweredige vloei na die sandfilter.



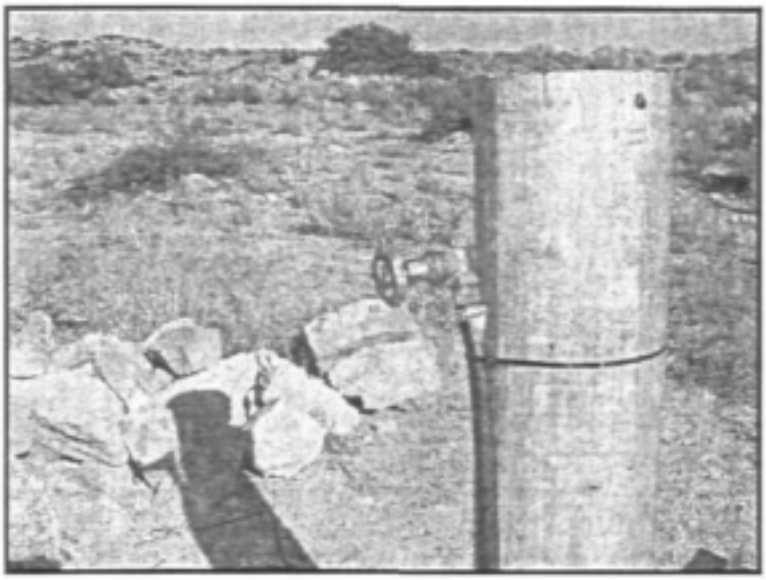
Riviersand: Riviersand groter as 2mm. Word gebruik om die sandfilter volume op te maak vanaf die growwe sand tot 25mm onder die oorloop. Word verkry wanneer van die omliggende riviersand deur 'n 2 vierkantmeter, 2mm fyn vlekvrye staal gesif word. Alle materiaal wat bo-op die sif agterbly word gebruik.

Growwe gruis: Padgruis (grote 5mm tot 10mm): Onderste laag van sandfilter tot 50mm bo sifvoering.

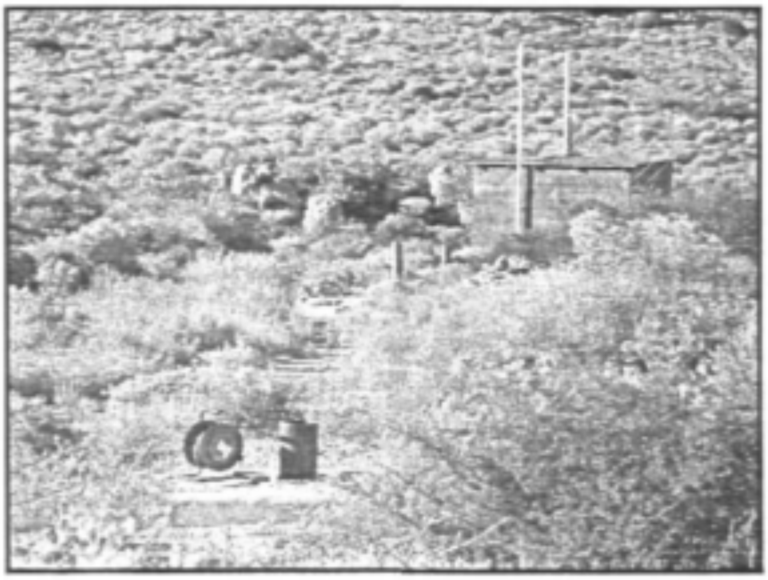
Inlynfilter: Vlekvrye staal sif filter in houer geïnstalleer, net na die vloeikraan in die aanvulllingsvoerpyp.

LDPE: Lae digtheid polietileen. 50mm swart aanvullingspyp.

Luguitlaatkraan: 15mm kraan gekoppel aan die aanvullingsboorgat wat enige opgeboude lugdruk in die aanvullingsvoerpyp sal uitlaat om die watervloei te verbeter. Moet toegemaak word tydens aanvulling.

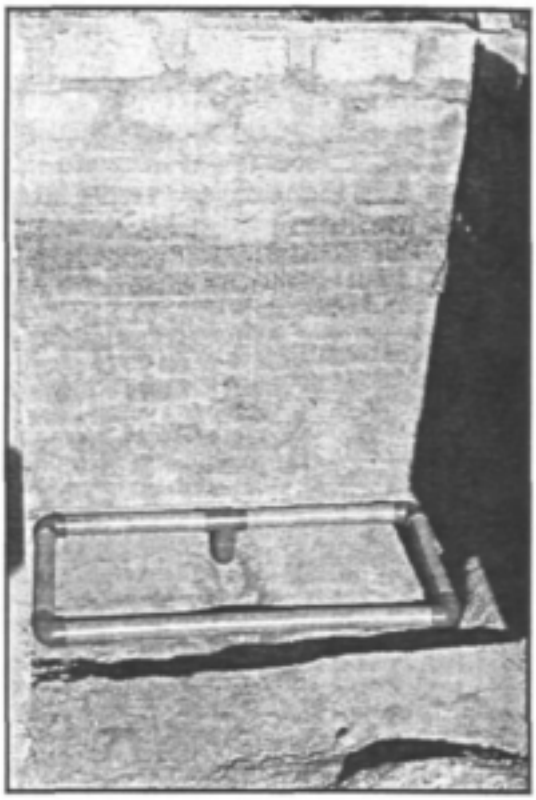


Moniteringsboorgate: Boorgate G45755, G45756 en G45758. Geboor om die grondwatervlakke te moniteer en om kunsmatige aanvulling te meet.



Produksieboorgat: Boorgat G39002 binne in die baksteen gebou wat met 'n dompelpomp toegerus is.

Sifvoering:	Blou, 63mm, Klas 6, 0.3mm PVC pyp met 0.3mm perforasie en gekoppel aan die sandfilterafvoerpyp.



Sifvoeringstutte:	Kort lengtes 50mm LDPE pyp onder die sifvoering geplaas as stutte. Dit sal verhoed dat die sifvoering se aanhegpunt met die sandfilterafvoerpyp breek.

Growwe sand:	Ligte kleur kwartsagtige 16/40 filtersand. 100mm laag bo-op growwe gruis.
Vloeirigtingskleppe:	Vlekvrye staal koeëltipe kleppe wat die filtervloeirigting beheer tussen die produksie- en aanvullingsboorgate.
Vloeikraan:	Kraan by die eerste T-aansluiting met die filtervoerpyp wat vloei na die vloeirigtingskleppe beheer. Word toegemaak tydens skoonmaak take.

Karkams Kunsmatige Aanvullings Skema: Boorgat Opnames

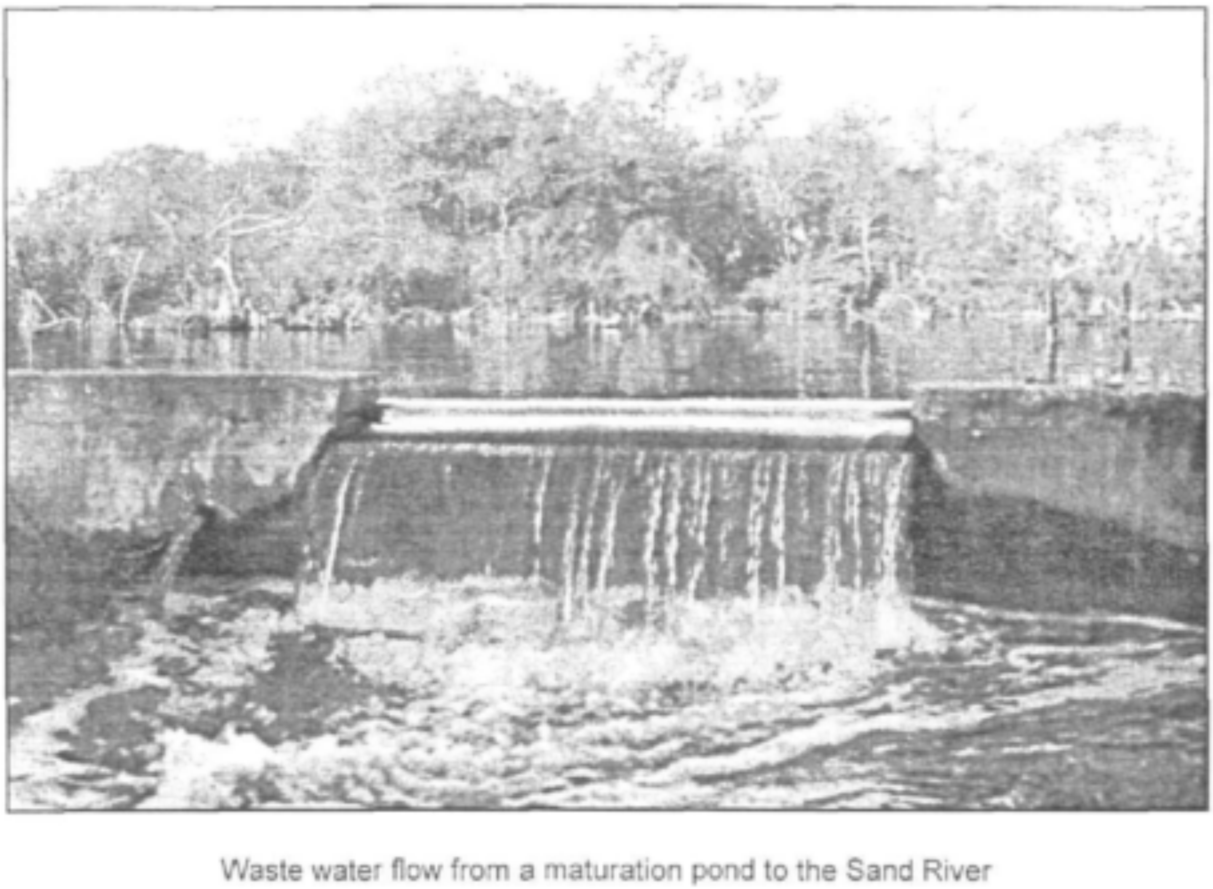
Datum	*Tyd*	*Watervlak G45756 (meter onder bekhoogte)*	*Vloeilesing geneem tydens : Pomp (P); Aanvulling (A); of Rus (R)*	*Onttrekkings-vloeimeter lesing (G39002) (kL)*	*Aanvullings vloeimeter lesing (kL)*	*Reenval (mm)*	*Opmerkings (Noteer kunsmatige aanvullings begin datum; skoonmaak van die filter ; monsterneming episodes; staking van aanvulling; hernuwingsdatum van filter na aanvulling; (ens)*

SECTION 4:

Polokwane Artificial Recharge Study

by

I Saayman, E C Murray & G Tredoux



Waste water flow from a maturation pond to the Sand River

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1. INTRODUCTION

Polokwane and Seshego in the Northern Province has in recent years experienced large population and economic growth. In most years the town's surface water sources are able to meet local needs. The town also has an elaborate groundwater abstraction infrastructure that is able to supply groundwater for domestic use during peak demand periods as well as in times of surface water shortages.

Well fields are located along the banks of the Sand, Blood and Sterkloop Rivers. The present Sand and Blood River groundwater abstraction system is designed to utilise water from predominantly the deeper secondary aquifer. It is suspected that a proportion of the water abstracted from the Sand and Blood River well fields is derived from the infiltration of water that is discharged from local Waste Water Treatment Works (WWTW) into the Sand River. In effect it would mean that the treated wastewater is artificially recharging the aquifers supplying the town, which in effect leads to an indirect recycling of the treated wastewater. As a result some concern exists on the longer term impact that waste water recharge could have on the quality of water in these aquifers.

1.1 Aims and Objectives

This study aims to broaden the understanding of the hydraulic system in the area of groundwater abstraction north-west of Polokwane. For this reason the Sand River groundwater abstraction area was selected for study. The similarities in the Sand and Blood River groundwater systems means that the findings and conclusions derived from study of the Sand River site could be applied to the management of the Blood River groundwater abstraction site.

The present study focuses on the alluvial deposits along the Sand River, from the Polokwane WWTW to the river's confluence with the Blood River. Although it is suspected that WWTW discharge results in recharge of the local aquifer systems, this has not been systematically studied and documented. This study aims to address this uncertainty by answering the following questions:

1. Is there a hydraulic connection between the upper and lower aquifers, and if so, how much water from the upper aquifer is recharging the lower aquifer?
2. If the two aquifers are hydraulically connected, what is the impact on the quality of water in the lower aquifer?

1.2 Method of Study

The following steps were taken to achieve the stated aims:

- A literature review provided an overview of the theory of Artificial Recharge infiltration schemes;
- Existing water quality and water level data were reviewed and interpreted;
- New boreholes were drilled and test pumped;
- The resultant test pump data was analysed and interpreted;
- The results of the studies were used to derive conclusions;
- Report compilation.

To a large extent the study relied on the review of existing literature and the collation and interpretation of chemistry and flow data (see Table 1). A number of reports were received from Polokwane municipality and the Northern Province regional office of the Department of Water Affairs and Forestry (DWAF). Among the reports reviewed were:

- Reports compiled by Water System Management for Polokwane Municipality (WSM, 1995, 1997a, b, c)
- A joint Ninham Shand-Water System Management report (Ninham Shand and WSM, 1995)

The project further involved the interpretation and analysis of chemistry data. Data from the following sources were collected and analysed:

Table 1 Chemistry and volume data sources interpreted during study.

Data Description	*Source*	*Period*
Sand River water quality	DWAF (NWQD)	Mar. '95 – Aug. '99
Borehole water quality	Polokwane Municipality	Mar. '89 – Jun '99
Borehole water quality	WSM	'94 -'95, '97 - Dec '99
WWTW discharge quality	Polokwane Municipality	Jan '96 – Dec '98
WWTW discharge volume	Polokwane Municipality	Jan '96 – Dec '98
Sand River flow volumes	DWAF (NWQD)	Mar. '95 – Aug. '99
Monthly precipitation	National weather service	Jan. '95 – Dec. '98

In addition new chemistry and pumptest data were collected and analysed. The method used and the data gathered during the test pumping is described in section 4.

2. ARTIFICIAL GROUNDWATER RECHARGE

2.1 Direct Surface Recharge

Direct surface recharge techniques such as bank filtration and basin filtration are among the simplest and most widely applied artificial recharge methods. In this method, water moves from the land surface to the aquifer by means of percolation through the soil. Most of the existing large-scale artificial recharge schemes make use of this technique, which typically employs infiltration basins to enhance the natural percolation of water into the subsurface.

Field studies of infiltration techniques have shown that, of the many factors governing the amount and quality of water that will enter the aquifer, the area of recharge basin and the length of time that water is in contact with the soil is the most important (Todd, 1980). Some of the arguments for recharge of water to aquifers over surface storage include: the ability of water infiltration to eliminate pathogens, its ability to remove chemical pollutants by natural means, its attenuation of quality fluctuations, and the benefit of storage in a medium largely protected from atmospheric fall-out (Stuyfzand, 1998).

2.2 Design of infiltration schemes

During the design and planning of artificial groundwater recharge projects the following basic factors have been identified as important: the location of geologic and hydraulic boundaries, depth to the aquifer and lithology, storage capacity, porosity, hydraulic conductivity, and natural inflow and outflow of water to the aquifer; the availability of land, surrounding land use and topography; the quality and quantity of water to be recharged; the economic and legal aspects governing recharge; and the level of public acceptance (Todd, 1980, Murray and Tredoux, 1998, Stuyfzand, 1998).

The two key hydrogeologic factors for the successful implementation of an infiltration scheme are:

1. The ability of the aquifer to receive the recharge water (i.e. its storage and permeability characteristics). There are two main physical characteristics that determine whether an aquifer is suitable for accepting, storing and allowing for the recovery of artificially recharged water. They are the aquifer's hydraulic conductivity and storage capacity. A third important factor is the aquifer's hydraulic gradient, which relates mostly to the recovery of the recharged water. Aquifers which have high hydraulic conductivities and which have high storage capacities are more suitable for receiving additional recharge water than those with low hydraulic conductivities and low storage capacity. However, if the aquifer has a high hydraulic conductivity and a high hydraulic gradient, water will flow rapidly away from the point of recharge and may be difficult to recover (Murray, and Tredoux, 1998)

2. The quality of the recharge water should be such that clogging of the infiltration

surface is minimised or prevented, particularly with regard to suspended material and turbidity. It is also important that the recharge water be chemically compatible with the aquifer material through which it flows and the naturally occurring groundwater for avoiding chemical reactions that would reduce aquifer porosity and recharge capacity (Murray and Tredoux, 1998). Contaminants should not be present in the recharge water unless they can be removed by pre-treatment or chemically decomposed by a suitable land or aquifer treatment system.

2.3 Soil Attenuation

The use of sand and soil as filter material has a generally positive impact on the quality of the infiltrating water (Shelef, 1990, Icekson-Tal & Blanc, 1998, Kivimäki, *et al.*, 1998, Sililo, *et al.*, 1999). During infiltration, attenuation processes may take place in both the unsaturated and the saturated zones of the soil column. Such treatment includes the processes of filtration that occur in the top layer of soils; physicochemical processes such as adsorption, ion exchange and precipitation that occur throughout the soil profile; and biological processes such as the breakdown of organic matter and full nitrification and partial denitrification, which occur both in the unsaturated and saturated zones (Sililo, *et al.*, 1999).

However, not all subsurface environments are equally effective in their attenuation properties. The degree of attenuation is to a large extent dependant on the properties of the porous media. The properties that are important in contaminant attenuation include surface area, particle size, structure, mineralogy, organic and mineral coatings (Sililo, *et al.*, 1999).

At least one study has shown that natural bank filtration can result in Total Organic Carbon (TOC) and Assimilable Organic Carbon (AOC) reductions of up to 40% and 72% respectively (Kivimäki, *et al.*, 1998). The reductions in organic carbon were shown to occur mostly in the littoral zone. The Kivimäki, *et al.* (1998) results further suggest that in saturated, reducing conditions only minor microbial degradation will occur.

Filtration accounts for the removal of much of the particulate material and turbidity, and to a certain extent the removal of micro-organisms, while natural die-off accounts for additional microbial removal. Chemical processes are responsible for the removal of phosphorous (mainly precipitation) and some trace elements (Cr, Cd, Cu, Mo, Ni and Se), while biological processes improve COD/BOD removal, and are responsible for nitrification and denitrification (Sililo, *et al.*, 1998, Icekson-Tal & Blanc, 1998). The use of sand filtration has been shown to make a valuable contribution to the integrated management of water resources by addressing the dual problems of wastewater discharge and water shortage (Shelef, 1990, Icekson-Tal & Blanc, 1998).

3. STUDY AREA DESCRIPTION

3.1 Location

The study site is located north-west of Polokwane, within the alluvial deposits of the Sand River, between the Polokwane wastewater treatment works and the confluence of the Sand and Blood Rivers (Figure 1). The topography of the study area is fairly flat, with a gradual decrease in elevation from south to north. The area lies between an elevation of 1180 mamsl and 1300 mamsl. The riverbanks support a thick vegetation cover of trees and grasses extending up to approximately 350 meters from the river channel.

Figure 1 Study site with borehole location and layout. [source: WSM, 1994]

3.2 Climate

The local climate is strongly influenced by the region's elevation above sea level and its latitude, just south of the Tropic of Capricorn, which combine to give an average daytime maximum temperature of 27°C in summer and 20°C in winter. Summer rainfall varies between 400-600 mm per year, with a mean annual precipitation of 591 mm (Orpen, 1986). Figure 2 shows that evaporation exceeds precipitation throughout the year.

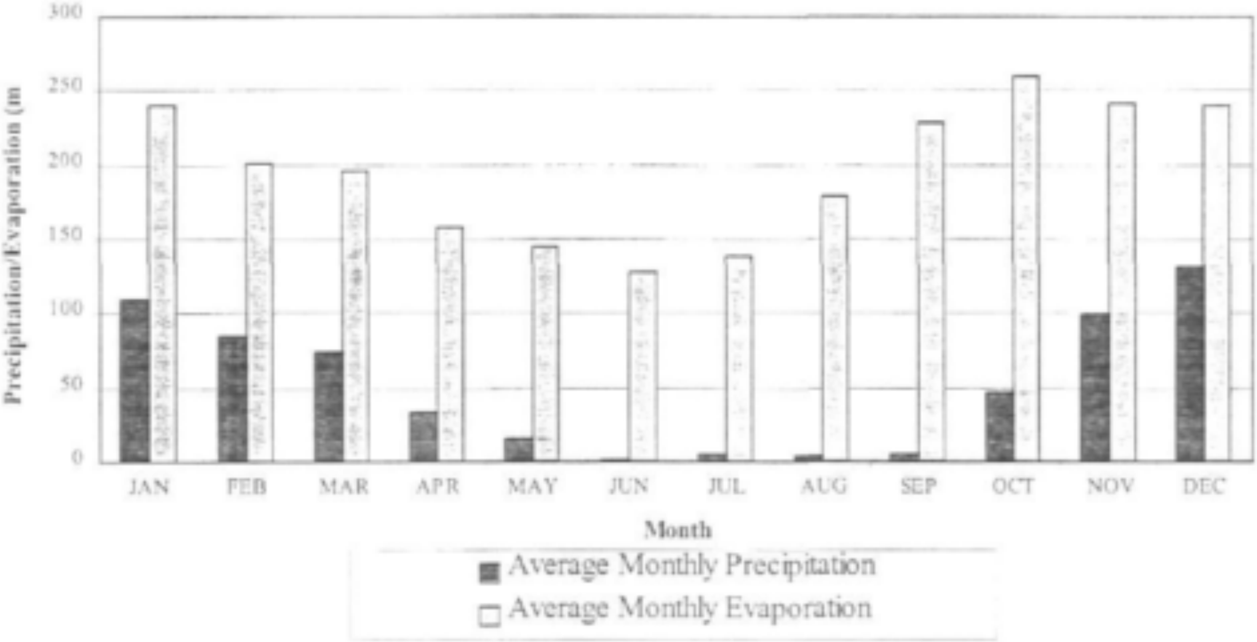

Figure 2 Average monthly precipitation and evaporation in the Polokwane area. (Station: 0677802A5 - Polokwane - WK: 1993/01/01 - 1998/12/11)

3.3 Water Supply Infrastructure

The Polokwane and Seshego municipalities have supplemented their surface water supplies with two well fields from as far back as 1965. An extensive study on the supply of groundwater to the town of Polokwane was commissioned and completed in 1985 by DWAF. The study included extensive geophysics, some exploration drilling, aquifer test pumping and sampling. It was concluded that it is possible to locate boreholes in the gneiss aquifer with yields in excess of 15 l/s. Based on this, the expansion of the town's groundwater abstraction infrastructure was proposed.

4. GEOLOGY AND HYDROGEOLOGY

4.1 Geology

Most of the study area is covered by alluvium, which extends to about 300 m either side of the of the Sand River drainage channel, reaching depths of up to 25 m. The sand deposit consist of upper clayey sands overlying coarser sands, and gravel boulder layers near or at its base. The sandy gravel layers are the most permeable part of the alluvial aquifer. A contour map of sand thickness is presented in Figure 3 (Ninham Shand, and WSM, 1995).

Underlying the alluvium deposits are the granite gneiss rocks of the Hout River Gneiss Complex. The granite gneiss rocks are recognized as the oldest of the intrusive phases within the Pietersburg Group. The granitic gneisses have probably been subjected to various phases of granitization. Pegmatitic and amphibolitic bands are common. Weathered basins and fractured gneisses form permeable zones to a depth of 60 meters (WSM, 1997).

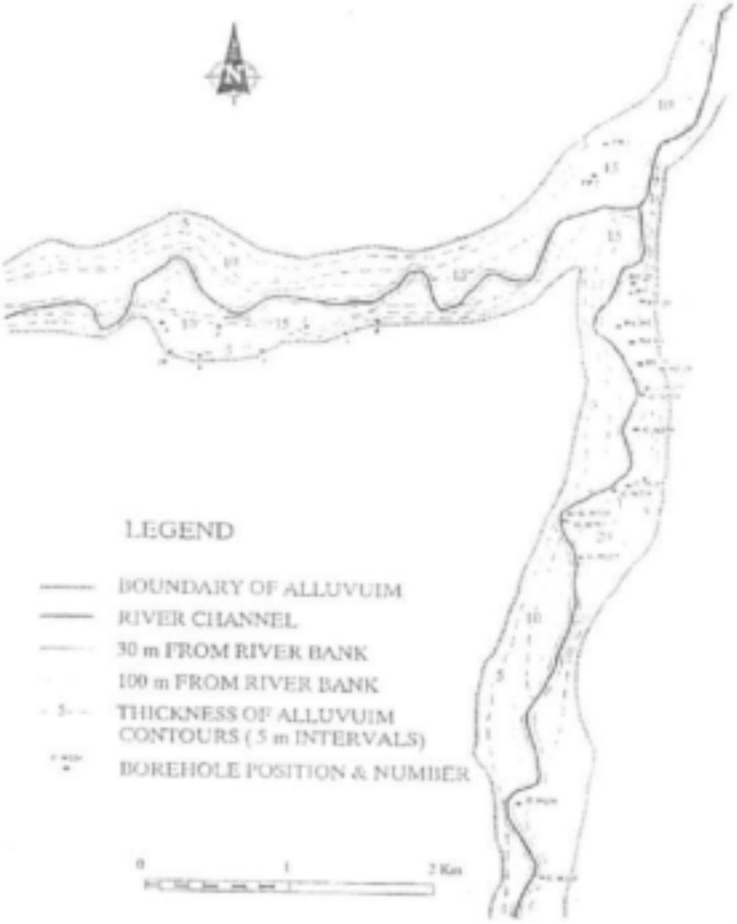

Figure 3
Contours of sand thickness along the Sand and Blood Rivers. [Source: Ninham Shand, and WSM, 1995]

The granite gneisses are heavily intruded by dolerite dykes (see Figure 4). These dykes generally strike NE - SW and are associated with Karoo age volcanics. It is postulated that the dolerite dykes in places enhance fracturing and weathering of the adjacent gneisses (WSM, 1997). It should also be considered that the dolerite dykes may have the effect of compartmentalising the aquifers within the granite gneiss. Fracturing on lithological contacts with the host rock, on the margins of intrusive dykes and through various processes of granitization have probably enhanced the gneiss's capacity to store water (Orpen, 1986). The permeability of the granitic gneiss generally decreases with increasing decomposition. This may in places lead to the development of a clay layer between the partly decomposed granitic gneiss and fresh bedrock.

Fracturing in the gneisses may not necessarily only be vertical, but may be horizontal as well. Such sheet joints in gneisses generally result from decompression that occurs when erosion has removed a considerable thickness of rock. Examples of these are the sheet joints that developed in the Archean shield of southwestern Australia. Here the joints are sub-parallel to the existing natural surface and are often very open, as much as 10 cm (Brown, *et al.*, 1983). These extensive sheet joints occur one beneath another in the solid rock to a depth of 100 m or more. The deep joints are less open than the shallow joints, because of increased pressure with depth. The shallow joints are known to be capable of storing and allowing movement of groundwater to depths of at least 25 meters (Brown, *et al.*, 1983). The porosity and permeability of gneiss rocks are generally very low. Porosity in such rocks generally vary between 0.1 to 3 %, but may increase through weathering to 50 % (Brown *et al.*, 1983).

The permeability of the bedrock is a function of the fractures found in the rock. The permeability of metamorphic rocks, such as gneisses for instance, tend to be highly directional. Studies have shown that fractures in such rocks, perpendicular to the schistosity or cleavage, give more water than oblique fractures, and oblique fractures give more water than parallel fractures (Brown, *et al.*, 1983).

4.2 Hydrologic Units

Three water units are recognised within the study area.

- Surface water, which represents overland flow derived from upstream in the main catchment, discharge from the wastewater treatment works, and base flow discharging from the groundwater system.
- Groundwater in the upper alluvium aquifer, recharged from precipitation and in interaction and exchange with surface water in the river channel and groundwater in the deeper granitic gneiss aquifer.
- Groundwater in the deeper granitic gneiss aquifer, recharged from precipitation and the alluvium aquifer.

In understanding the hydraulic system within the study site it is important to recognise that a hydraulic connection between surface water in the Sand River and groundwater is plausible. Where both groundwater and surface water form part of the hydrological system and they are not separated by impermeable layers, interaction may occur. This may either take the form of groundwater discharge to the stream (a gaining stream) or groundwater recharge from the stream (a losing stream). A stretch of river may be both gaining and losing at different times of the year depending on water levels in the stream and in the surrounding groundwater system. As these water levels are dependent on the input of water to the system (usually derived from precipitation and runoff) the nature of the surface water-groundwater interaction may vary with the seasons.

However, the nature of flow in this section of the river is considerably altered by discharges from the Polokwane Waste Water Treatment Works (WWTW). Annually about 5 x 10^6 m^3 water is discharged from the WWTW (WSM, 1994), and this figure appears to have increased to 5.9 x 10^6 m^3 over the past few years (Table 4). The combination of groundwater abstraction and WWTW discharge, has resulted in the development of an undulating groundwater water-level surface. A groundwater mount appears to occur in the area immediately downstream of the WWTW. As a result the natural groundwater flow in the study area has been altered (WSM, 1994).

Orpen (1986) estimates a catchment-runoff yield of 20 000 m^3/km^2 for the Sand River quaternary catchment A511 (about 530 km^2). Surface water flowing in the Sand River will inevitably recharge the associated alluvial aquifer, and because of the hydraulic connection between them, also the deeper lying granitic gneiss aquifer. The volume of water that each of these systems are able to accept is difficult to estimate. (Some estimates on recharge volumes are made in section 4.7)

Using the figures quoted by Bredenkamp *et al.* (1995) for the Dendron area, where the local geology is very similar to that in Polokwane, an estimate for the amount of direct recharge to the Polokwane granite gneisses is made as follows:

Recharge as a fraction of Rainfall x Mean Precipitation = Average Recharge (mm/a)

0.015 x 591 = 9.1 mm/a

Although only a rough estimate, it does give some indication of the amount of water that could be recharged to the secondary aquifers if the granitic gneiss rocks received recharge from rainfall only. The water that is recharged to these aquifers through direct recharge is, however, expected to be much smaller than the recharge taking place through the alluvium deposits. It is conceptualised that leakage does occur between the alluvium sand aquifer and the fractured gneiss aquifers. The test-pumping operation was designed to verify this and to estimate the amount of leakage.

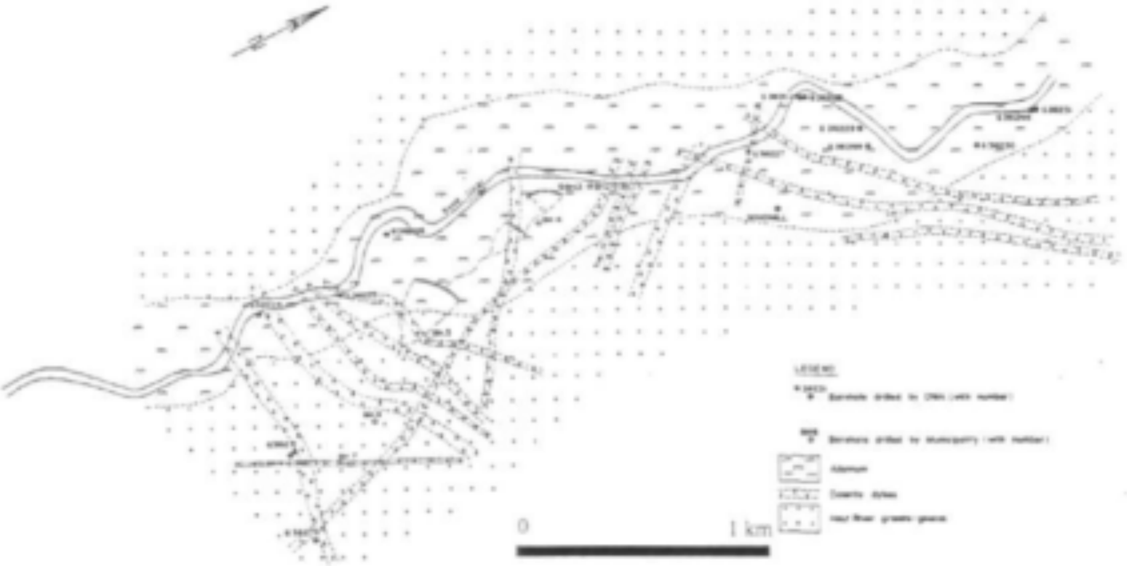

Figure 4 Local geology and location of government boreholes in the study area as mapped by DWAF [source: DWAF, 1986]

4.3 Groundwater Yields

Hydro-censuses conducted in the study area all confirm that boreholes drilled into the secondary aquifers in the granite-gneiss rock are high yielding (Orpen, 1986, WSM, 1995). Yields above 20 L/s are common, with a yield of 60 L/s recorded in one borehole (WSM, 1995). Many of these boreholes were, however, not test pumped, and the yields quoted represent estimates from blow yields following drilling. Where boreholes were test pumped, yields vary between 10 and 20 L/s, with recommended sustainable yields of between 360m^3/day and 720 m^3/day (WSM, 1997).

Six boreholes with a total combined sustainable yield of 2952 m^3/day were drilled within the study area during 1997, and are presently used to supplement the town water supply. The boreholes were constructed to tap both the pebble horizon in the alluvium and the deeper secondary aquifer. The higher yielding water strikes are almost always found in the gneisses.

4.4 Groundwater Flow

Water levels in the study area were recorded subsequent to the 1997 abstraction period. The results of those observations are presented in Figure 5. Note that a major limitation in the interpretation of these water level contours is the lack of data points outside the immediate area surrounding the river channel.

4.5 Test Pumping Program

Two sites were selected for test pumping. Both test sites are located along the banks of the Sand River, with Site 1 located upstream of the outlet of the WWTW, and Site 2 lower down the river not far from the Sand River's confluence with the Blood River. The location of the two sites are presented in Figure 5. None of the existing boreholes at these locations complied with the requirements of the test program. This meant that boreholes had to be constructed to test-pumping specifications. The following sections give an outline of the construction of the boreholes, the test pumping and the pumptest data interpretation.

4.5.1 Drilling Results

A total of eight boreholes were drilled. These boreholes were to serve the dual purpose of both pumping boreholes and observation boreholes during the pumptests. Six of the boreholes were drilled at Site 1 and two at Site 2. A summary of the drilling results is presented in Table 2.

Table 2 Summary of drilling results.

Bh. No.	*Depth (m)*	*Casing*	*Grout Seal (m)*	*Depth of Sand (m)*	*Depth of weathering (m)*	*Static Water Level (m)*	*Strike Depths (m) - main strike in bold*	*Blow yield (l/s)*
				Site 1				
S1B1	81	27	15-27	15	34	5	**40**,70	5
S1B3	30	18	11-18	15	28	5	18,**21**	2
S1B6	16	16 perforated	-	16	-	5	-	-
				Site 2				
S2B1	75	16	8-16	14	27	5	18,**26**	5
S2B2	12	12 perforated	-	12	-	5	**10**	1

The boreholes were drilled to varying depths, each targeting a different hydrogeological unit. The two deep boreholes, i.e. boreholes S1B1 and S2B1, targeted the fresh bedrock. To prevent the leakage of water from the alluvium and the weathered rock into the borehole during pump testing, the alluvium and dolerite layers were pressure sealed with grout. Borehole S1B3 of site one was drilled to the base of the weathered material. To prevent infiltration during pump testing the transition to gneiss was sealed off with grout. Boreholes S1B6 and S1B2 were drilled to the base of the alluvium. The borehole logs for each of the newly drilled boreholes are presented in Appendix 1.

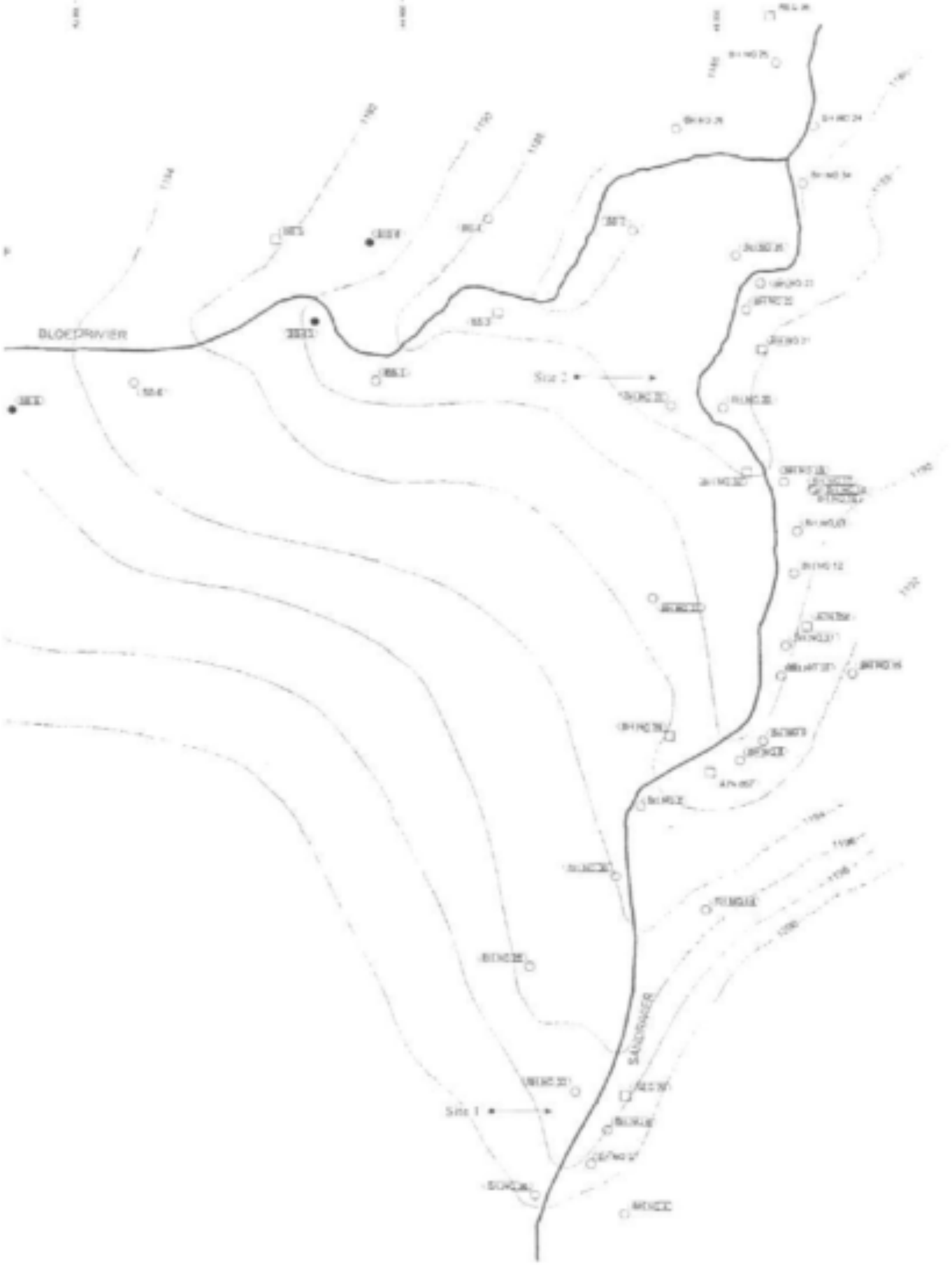

Figure 5 Location of study drill sites and water level contours from water level observations in surrounding boreholes (WSM, 1994)

4.5.2 Description of tests

The pumping tests were designed to establish whether a hydraulic connection exists between the upper alluvium, the layer of dolerite and the lower gneisses. The results of the test at Site 2 were discarded, as it is suspected that the alluvium was not totally sealed off in borehole S2B1. During pumping of this borehole, sand entered the pump, indicating that the borehole's upper seal was not effective.

The tests at site 1 were successful, and are presented below. Each borehole was pumped in order to establish the hydraulic relationship between the three units and to obtain hydraulic parameters for each unit. *The drawdown curves are shown in Appendix 2.*

4.5.3 Pumping borehole S1B1 (the deep, gneiss borehole)

Figures 1-3 of Appendix 2 show the results of the step test on pumping borehole S1B1. Water levels stabilised during the third step, which suggests that leakage occurs from the saturated sands. The hydraulic connection between the three lithostratigraphic units is evident from the water level response to pumping, in the observation boreholes (Appendix 2, Figures 2 & 3).

The constant discharge tests confirmed the hydraulic connection between the three units (Appendix 2, Figures 4 - 6). This test is a repeat of an earlier constant discharge test that was conducted at a rate of 6.7 L/s. During the 6.7 L/s test the water level stabilised at 11.5 mbgl from 10 minutes to 700 minutes. This flattening of the drawdown curve is typical of leaky aquifers. At the pumping rate of 10 L/s (Appendix 2, Figure 4), the leakage effects are only evident for a short period (from 12 to 40 minutes). After 40 minutes, the high pumping rate caused a rapid drop in the water level.

The early-time transmissivity value obtained from the pumping borehole was 32 m^2/day. This reflects the permeability of the high yielding fractures at this site. The late-time transmissivity value reflects the permeability of the more wide spread micro-fractures, and was found to be 5 m^2/day.

These relatively low transmissivity values mean that localised areas of dewatering should occur around production boreholes, and that during droughts, the aquifer around the pumped boreholes would be dewatered. However, historical data shows that this does not happen, rather, the boreholes maintain their production rates irrespective of droughts. The reason for this is because leakage is induced from the alluvium above, which in turn, is constantly recharged by the waste water treatment works.

Figure 5 (Appendix 2) confirms the hydraulic link between the three units. The rise in the water level in borehole S1B3 after 300 minutes is not due to a drop in the pumping rate. During the first constant discharge test at 6.7 L/s, this rise in the borehole water level was also observed - in this case, after 800 minutes of pumping.

The pumping of borehole S1B1 proves that there is a hydraulic connection between the three layers. Because the drawdown curves are not similar in shape to standard type-curves, conventional curve fitting methods could not be used to establish the vertical hydraulic conductivity (Kz).

4.5.4 Pumping borehole S1B3 (the intermediate, dolerite borehole)

Figures 6-11 (Appendix 2) show the results of the step and constant discharge tests on pumping borehole S1B3. The drop in water levels in the gneisses and the alluvium as a result of pumping from the dolerite confirms the hydraulic connection between these units.

The transmissivity value obtained for the dolerite is 2 m^2/day. Just like the drawdown curves obtained from pumping borehole S1B1, the drawdown curves from this test are not suitable for conventional curve fitting, and the vertical hydraulic conductivity could not be calculated.

The transmissivity of the dolerite is very low. If this dolerite is representative of the numerous dykes that cut through the study area, then these dykes will impede groundwater flow, and form a number of poorly linked hydraulic compartments.

4.5.5 Pumping borehole S1B6 (the shallow, alluvium borehole)

Figures 12 and 13 (Appendix 2) show the results of the short constant discharge test on pumping borehole S1B6. This test showed that abstraction from the alluvium induced a drawdown in the gneiss borehole.

The transmissivity value obtained for the alluvium was 30 m^2/day. This value was obtained from the recovery curve which gave a smooth, straight line (semi-log plot). The drawdown curve appears to have been affected by two boundaries, after 5 minutes, and again after 50 minutes. Considering that the vertical conductivity is usually less than horizontal conductivity, it appears as if the saturated alluvium serves as a slow, but steady source of recharge to the underlying gneisses.

4.6 Conclusions from the pumping tests

The pumping tests of the three boreholes that penetrate the three lithological units show that the units are hydraulically linked. The treated waste water that continually recharges the alluvium adjacent to the Sand River, is thus a source of recharge to the underlying gneiss aquifer.

4.7 *Numerical Modelling*

4.7.1 Conceptual Model

Based on the geological information (see sections 4.1 and 4.2), the aquifer was represented by three model layers: The borehole layout used during the modelling is presented in Figure 6. The following conditions were built into the model:

1. The pumping borehole S1B1 penetrated all three layers, but is sealed in the first and second layer.
2. A constant pumping rate of 10 L/s is used in the model.
3. The boreholes S1B3 and S1B6 were used to observe the water level fluctuations in the first and second layers respectively.

The positions of the three boreholes are shown below:

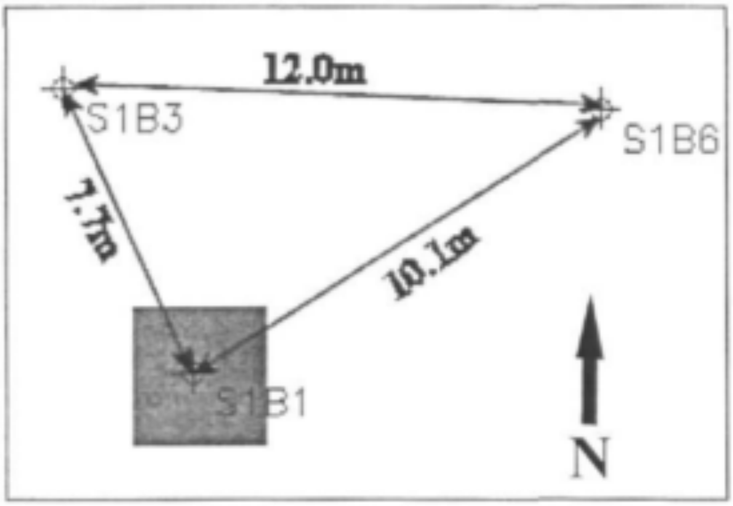

Figure 6 Borehole layout used in computer model.

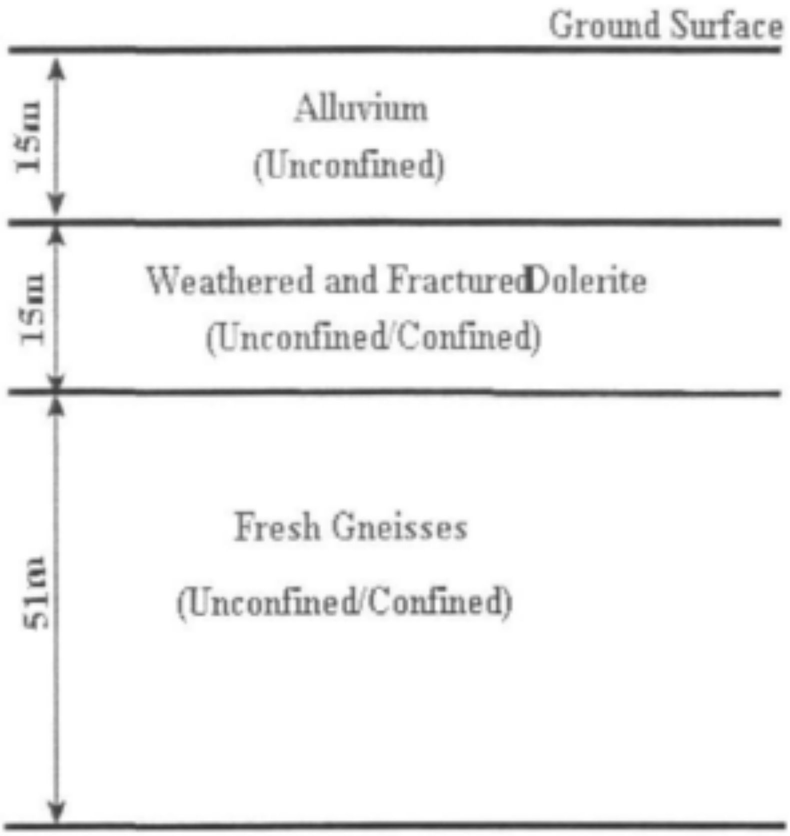

Figure 7 Conceptual model of model site geology.

4.7.2 Boundary Conditions

There are no special boundaries (constant head boundary, constant flux boundary, etc.) that exist in the study area, so the model domain was extended to a large area to make sure the drawdown induced by the pumping at the boundaries can be ignored. This allowed the use of no flow boundaries.

4.7.3 Numerical model

The Model domain is divided into 100 by 100 cells, each representing a size of 3 by 3 meters.

4.7.4 Model Results

By using the numerical model, the following parameters can be obtained.
Kh1 = 2.88 m/day (Kh means horizontal hydraulic conductivity)
Kh2 = 0.14 m/day

Kh3 = 0.29 m/day
Kv1 = 0.0086 m/day (Kv means vertical leakance)
Kv2 = 0.0058 m/day
Sy1 = 0.01 (Storage yield)
Sc2 = 0.004 (Storage coefficient)
Sc3 = 0.023

The Kv values obtained represent the vertical leakage from the sands to the gneisses. Assuming that the wetted front occupies, on average, an area of 300 m by 6000 m, then the annual leakage range, using the Kv values obtained, is 3.8 to 5.7 Mm^3. The lower value is considered to be too conservative because it is affected by the low-permeability dolerite layer. This layer is not continuous - rather, dolerite dykes, although common, do not occupy much of this area (Orpen, 1986). The lower value, which is 64% of the WWTW's annual discharge does however appear to be a reasonable figure. The higher value, which represents 97% of the WWTW's annual discharge is, on the other hand, considered to be too high. Possibly the figure lies somewhere in between these two values. The average of the two values (0.0072 m/day), gives an annual infiltration of 4.7 Mm^3, or 80% of the WWTW's annual discharge.

4.7.5 Assumptions

In the numerical model, the weathered and fractured dolerite is conceptualised as a horizontal layer and going through the whole model domain. Our understanding of the geology however indicates that a number of geological dykes transect the study area. The model is thus not necessarily a true reflection of the hydrogeologic conditions, however, it does give an indication of the leakage that could occur from the alluvium to the gneisses.

4.8. Water Balance Considerations

It has been proposed (see Appendix 2 of the main report for recommendations by Peter Dillon) that an estimate of groundwater recharge be derived by balancing the amount of water in the system and the amount of water that leaves the system through evapo-transpiration. In order to determine the rate of evapo-transpiration, a factor of 0.5 to 0.7 of pan-evaporation is proposed. This is to be multiplied by the area over which evaporation from the water surface and the alluvium aquifer takes place.

No detailed analysis of the wetted area surrounding the Sand River has been done, so, for the purposes of this report, a rough estimate of this area has been made. The estimate is complicated by the fact that about 2 kilometres downstream of the WWTW discharge point, the Blood River flows into the Sand River. The area is wetted downstream of the confluence, but there is contribution to flow from the Seshego WWTW into the Bloed River. For the purposes of our calculation we have estimated, in consultation with Mr C Haupt from Water Systems Management, that the wetted area is in the region of 1.5 Mm^2.

The amount of evaporation lost from the system can then be calculated as follows:

Average annual pan-evaporation in the study area is 2351 mm (or ~2.35 m).
Multiplied by the factor suggested by Dr. Dillon (0.5 to 0.7) to derive an evapo-transpiration figure of 1.18 m to 1.65 m.

The volume of water lost through evapo-transpiration is thus estimated to be 1.5 Mm^2 x (1.18 to 1.65 m) = 1.8 to 2.5 Mm^3.

The average annual discharge from the WWTW is 5.9 Mm^3.

The amount of the WWTW discharge that is lost to evapo-transpiration in an average year (if dry-season evaporation losses are assumed to occur throughout the year) is thus about 30 - 42% of the total.

If we were to assume that no other losses occur from the system, it would mean that 58 - 69 % of the discharge from the WWTW (or 3.4 - 4.1 Mm^3) is reused through either direct abstraction from the river, abstraction from the alluvium, or abstraction from the gneiss aquifer.

With the available information it remains impossible to determine the individual components of the system. However, from rough water balance calculations, it appears as if ~ 3.4 - 4.1 Mm^3, or 58 - 69 % of water discharged from the WWTW is available for recharging the gneissic aquifer.

5. WATER QUALITY ASSESSMENT

To a large extent this study largely consisted of a review of existing literature on the study site. The following tasks were complete as part of this study:

1. Literature overview of artificial recharge infiltration schemes;
2. A study of existing water quality and water level data;
3. The gathering and analysis of new water quality data;
4. The drilling and test pumping of new boreholes.

5.1 Historical Observations

Historical water quality data has been recorded on an ad hoc basis since the Polokwane well fields were put into production in 1965. Present data sets record water quality in the Sand River and Blood River well fields from 1989. The chemical analysis performed over this period focused on some selected anions and cations. However, no standard set of test parameters were used. The strategy also followed no continuous sequence of sampling, with long gaps of up to 3 years between sampling runs. The data generally lacks a continuous record of potassium, which may have proved a useful tracer for the tracking of discharges from the WWTW.

The lack of a clear sampling strategy and timetable, and the lack of a standard set of parameters tested, complicates the interpretation of the hydro-chemical data.

5.2 Background Water Quality

5.2.1 Sand River Water Quality (Natural Flow)

The natural quality of water in most river valleys in humid to sub-humid regions tend to be good to excellent. The observed total dissolved solids content is usually less than 500 mg/L. In semi-arid and arid regions the natural dissolved solids content is usually higher, and tends to range between 500 to 1500 mg/L (Brown, *et al.*, 1983).

A gauging station upstream of the WWTW discharge has been operational in the Sand River since 1978. Some water quality data were obtained from the National Water Quality database, and is presented below (Table 3). It should be noted that flow in the Sand River is seasonal in nature, and that the data presented in Table 3 represent only observations made during periods of surface flow.

Table 3 Summary of surface water quality of water in the Sand River upstream of the WWTW (source: National Water Quality Database of the DWAF)

	pH	*EC*	*TDS*	*Ca*	*Mg*	*K*	*Na*	*T.Alk*	*Cl*	*F*	*Si*	*SO4*	*NO2/3*
Mean	7.9	36.3	267	21.6	13.2	5.3	28.9	121	26.8	0.27	7.4	21.1	0.18
Max.	8.8	71.2	506	37.5	30	9.3	66.2	213	57.4	0.36	11.4	40.8	0.652
Min.	7.2	6.6	56	5	2.3	2.0	4.6	26	5.2	0.18	3.5	3.9	0.007

The time sequences for some of the chemical parameters are presented in Figures 8 to 10. The chemistry data is plotted together with monthly precipitation records for the same period. It may be assumed that the long periods over which no chemistry data are available represents periods with very little or no flow in the Sand River. The irregular nature of the sampling means that the data only provide an indication of the variation in chemical composition of Sand River flow between 1995 and 1999.

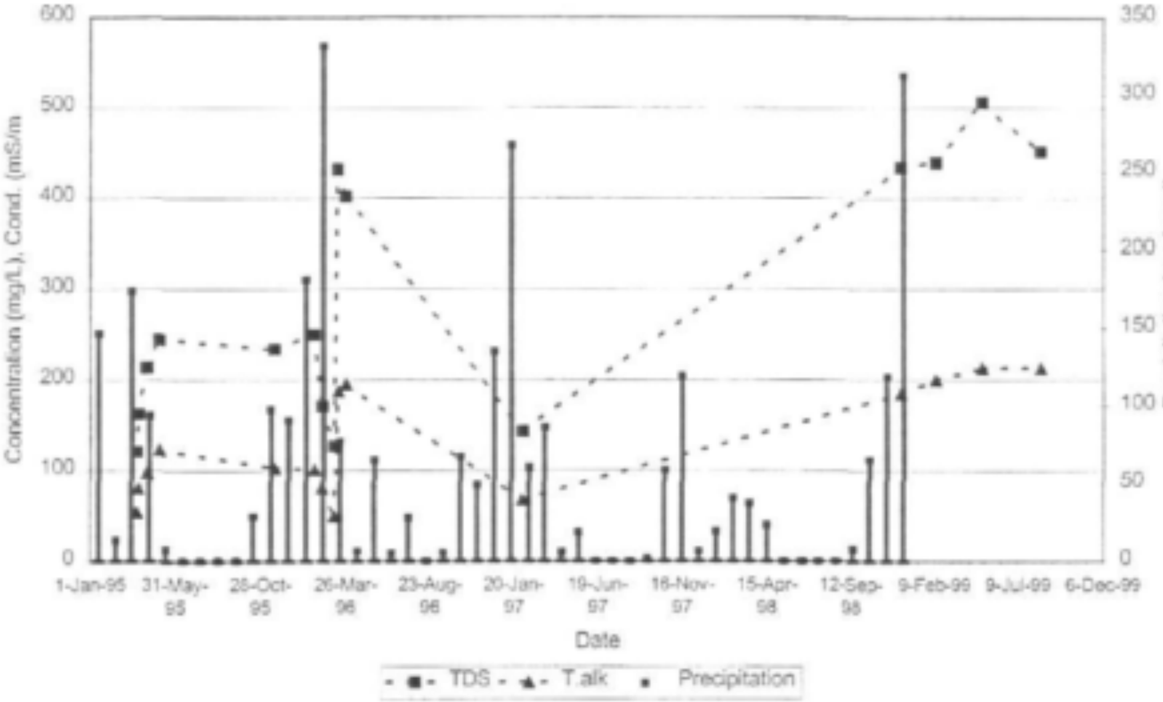

Figure 8 Water quality parameters electrical conductivity, sodium and chloride observed in water samples taken from the Sand River during flow, with monthly precipitation

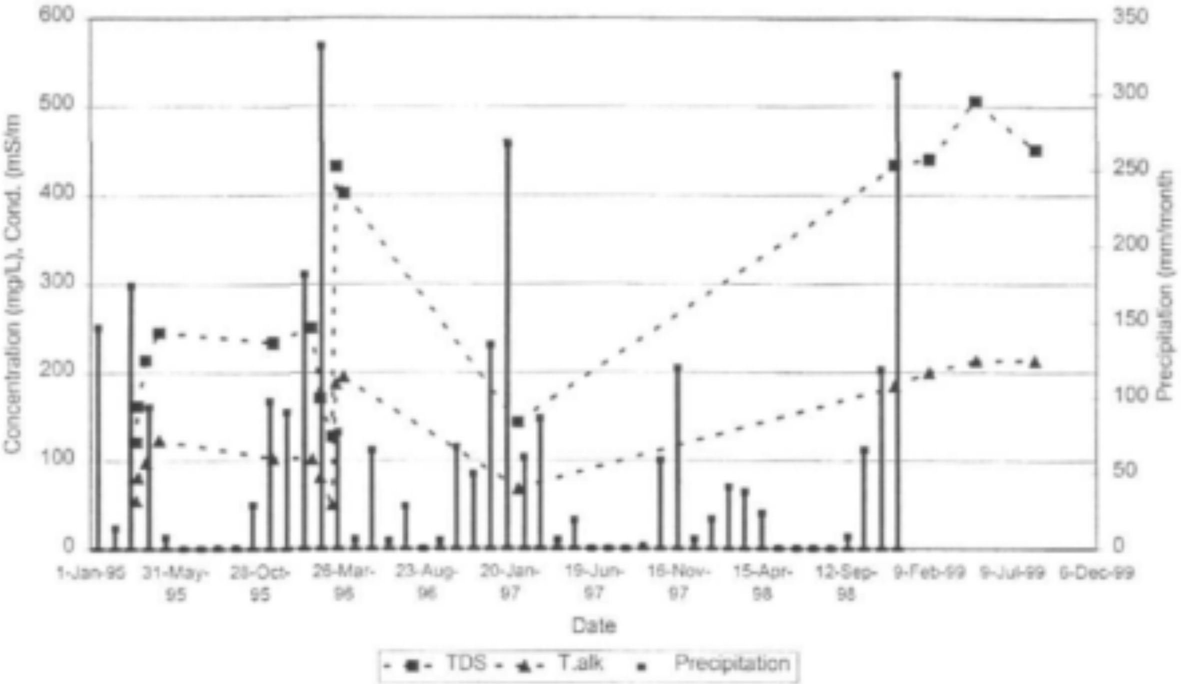

Figure 9 Water quality parameters total dissolved solids and total alkalinity in the Sand River during flow, with monthly precipitation

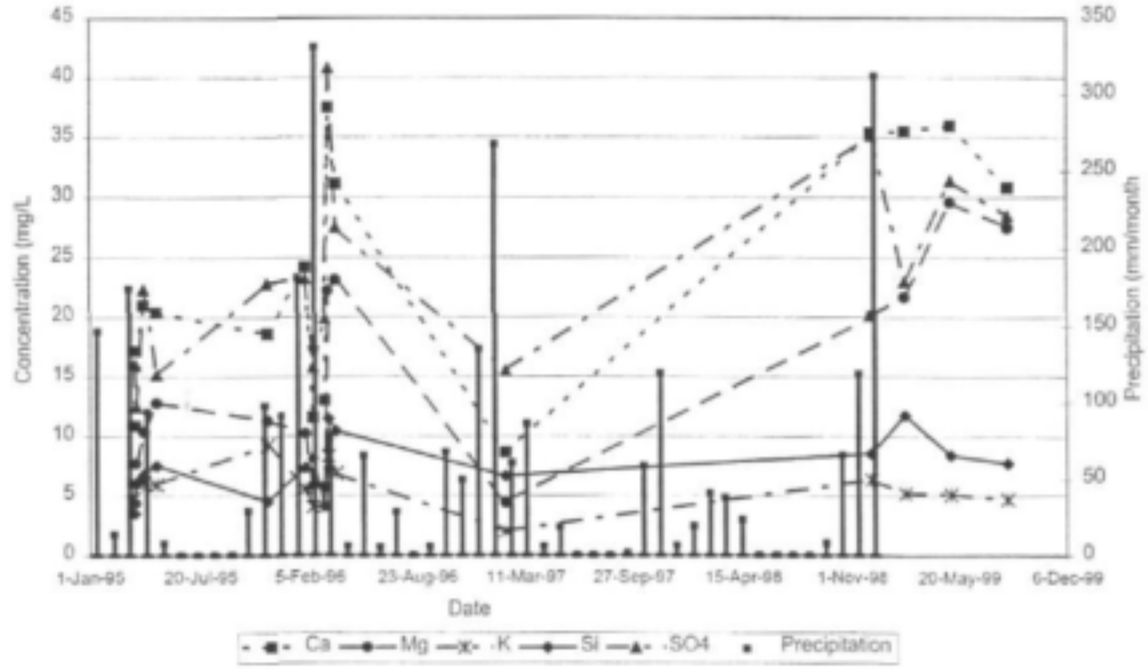

Figure 10 Water quality parameters calcium, magnesium, sulphate, potassium and silica observed in samples taken from the Sand River during flow, with monthly precipitation

Of interest is the way in which many of the parameters fluctuate, for example the observations between March and May 1995. Over this period electrical conductivity (14 to 33 mS/m), TDS (121 to 245 mg/L), sodium (8 to 20 mg/L), chloride (7 to 20 mg/L) and silica (4 to 8 mg/L) all show a steady increase in concentration. This deterioration in water quality could be the result of one of two processes, or a combination thereof. Firstly, it is likely that evaporation will result in an enrichment in the water's ion concentrations. Secondly, with the rainfall peak having been in March, it is possible that the later samples in this period represents a degree of groundwater discharge, with the enrichment in ion content the result of salt dissolution.

Down stream of the WWTW discharge point flow in the Sand River becomes perennial in nature, with the biggest contribution to flow for most of the year being made by discharge from the Polokwane WWTW.

5.2.2 Waste Water Treatment Works

The first sewage works was constructed at the Polokwane WWTW site in the 1950's. Over the past 50 years the treatment works have been expanded numerous times, with the last addition in 1996. This has increased the plant's treatment capacity to 19400 kL/d.

Water handled at the WWTW goes through both primary and secondary treatment. After which the waste water is channeled into a series of maturation ponds where retention is between 2 and 3 weeks. A mixture of waste waters are treated at the site, including domestic sewage and waste water from industries such as a local brewery, fruit juice manufacturer and an abattoir.

5.2.2.1 Discharge Volume

Records of the discharge volumes from the Waste Water Treatment works show little seasonal variation, with occasional peaks. The difference observed between the winter and summer discharge volumes is relatively small, with the larger discharge volumes in summer. A breakdown of the average discharge volumes is given in Table 4 and Figure 11.

Table 4 Polokwane Waste Water Treatment Works Discharge Volumes (1996 - 1998).

Average Daily Discharge	*16,251 m^3*
Avg. Daily Discharge (summer - Nov, Dec, Jan, Feb)	17,242 m^3
Avg. Daily Discharge (winter - Jun, Jul, Aug)	15,227 m^3
Avg. Monthly Discharge	494,606 m^3
Avg. Monthly Discharge (summer - Nov, Dec, Jan, Feb)	510,002 m^3
Avg. Monthly Discharge (winter - Jun, Jul, Aug)	466,342 m^3
Avg. Annual Discharge ('96 - '98)	5,915,472 m^3

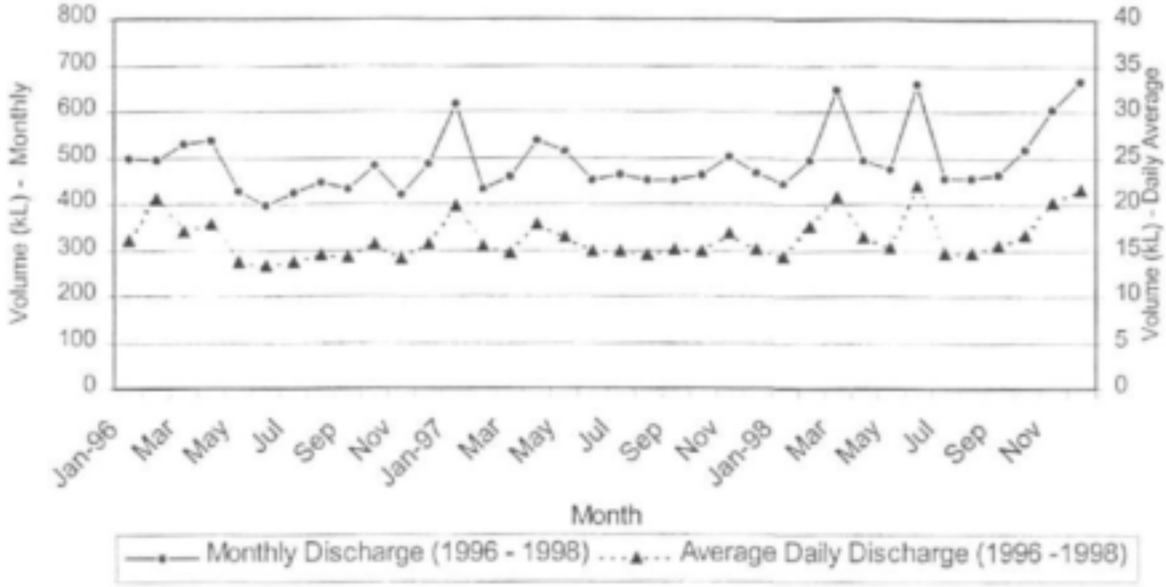

Figure 11 Volume waste water discharged from the Polokwane WWTW (kL) (1996 - 1998)
Source: Polokwane Municipality

5.2.2.2 Discharge Quality

Regular testing is done on the quality of water discharged from the Polokwane WWTW. Following treatment and retention in the maturation pond system the water is released into the Sand River (see Plate 3). Some of the key parameters are presented in Figures 12, 13 and 14. Some of parameters' concentrations occur over a wide range of values. The Total Dissolved Solids content for instance range between 275 mg/L and 1057 mg/l, and alkalinity between 133 and 387 mg/L. Other parameters are fairly stable and are found to fluctuate within a relatively narrow range (e.g. chloride). No clear seasonal trend could be observed in the fluctuations of the parameters.

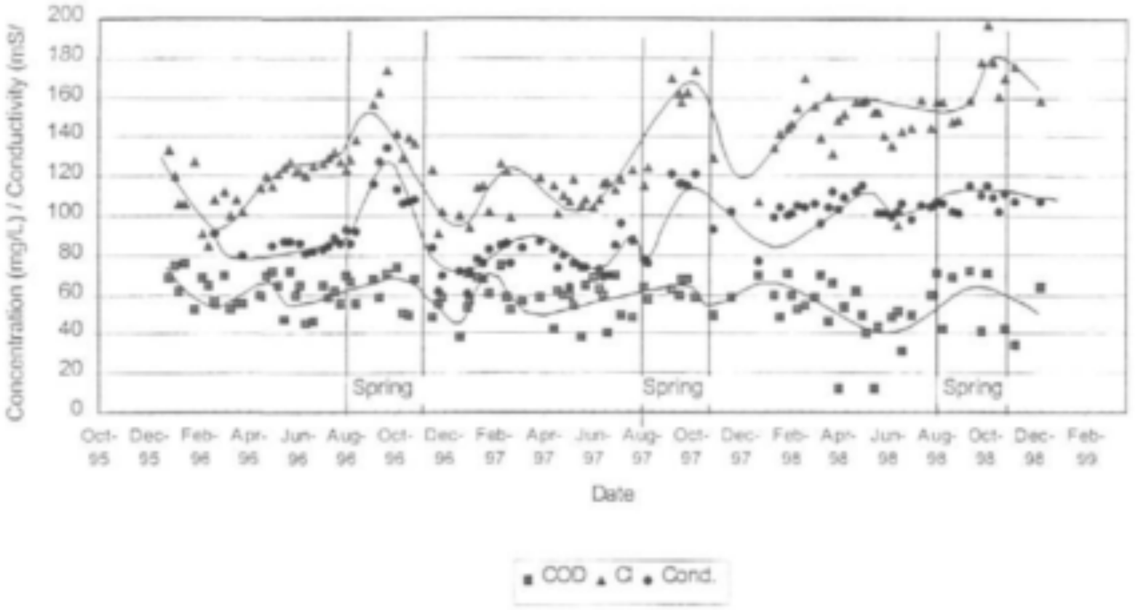

Figure 12 Polokwane WWTW discharge water quality (Chemical Oxygen Demand, Chloride, E.C.)

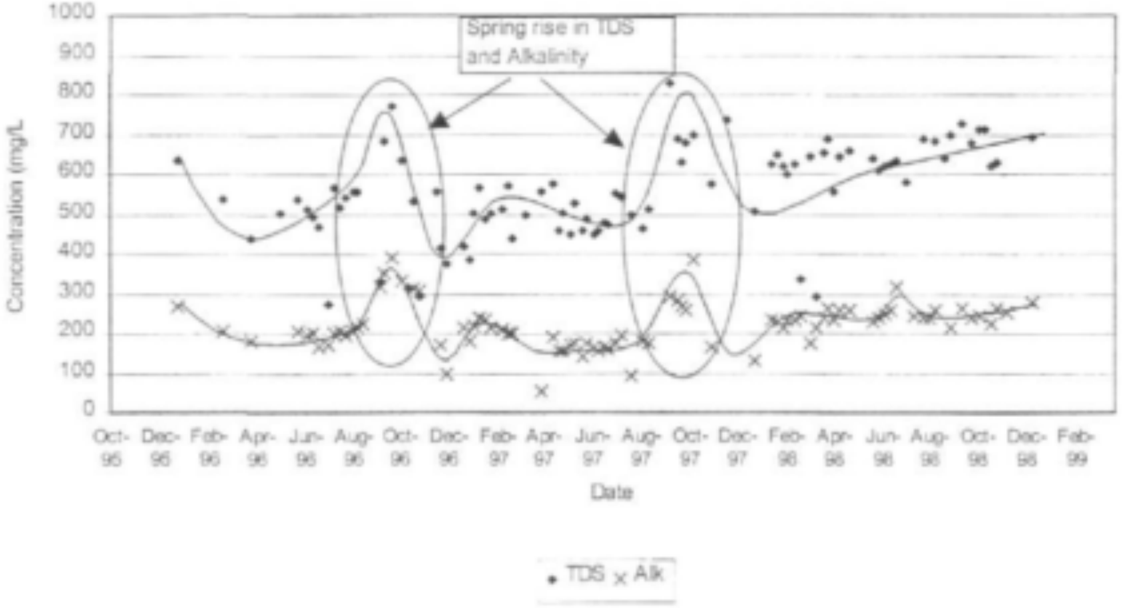

Figure 13 Polokwane WWTW discharge water quality (Total Dissolved Solids and Alkalinity)

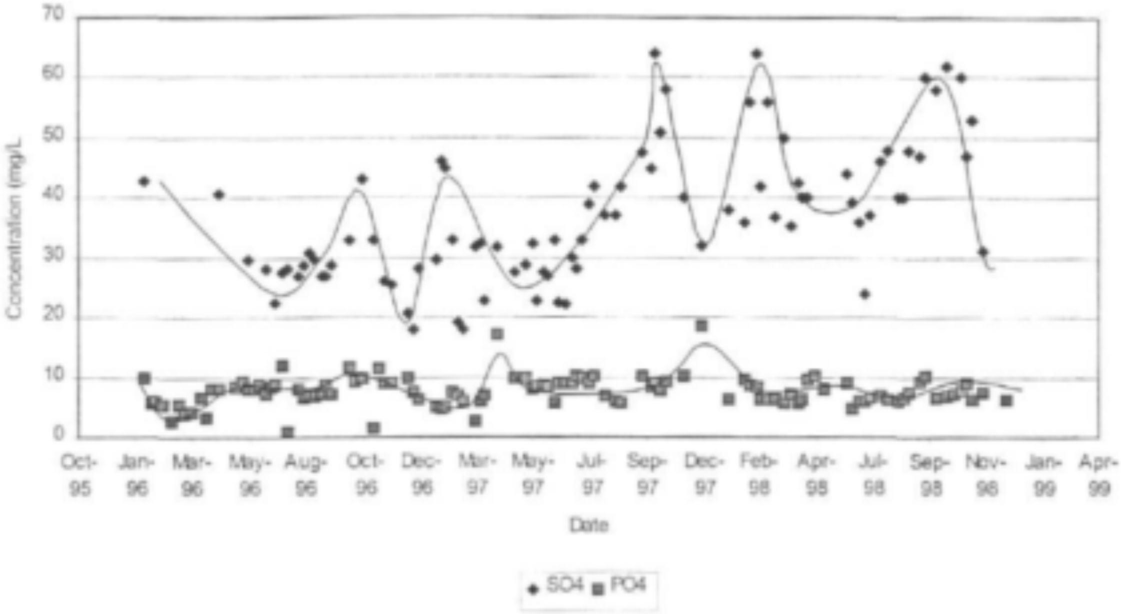

Figure 14 Polokwane WWTW discharge water quality (Sulphate and Phosphate)

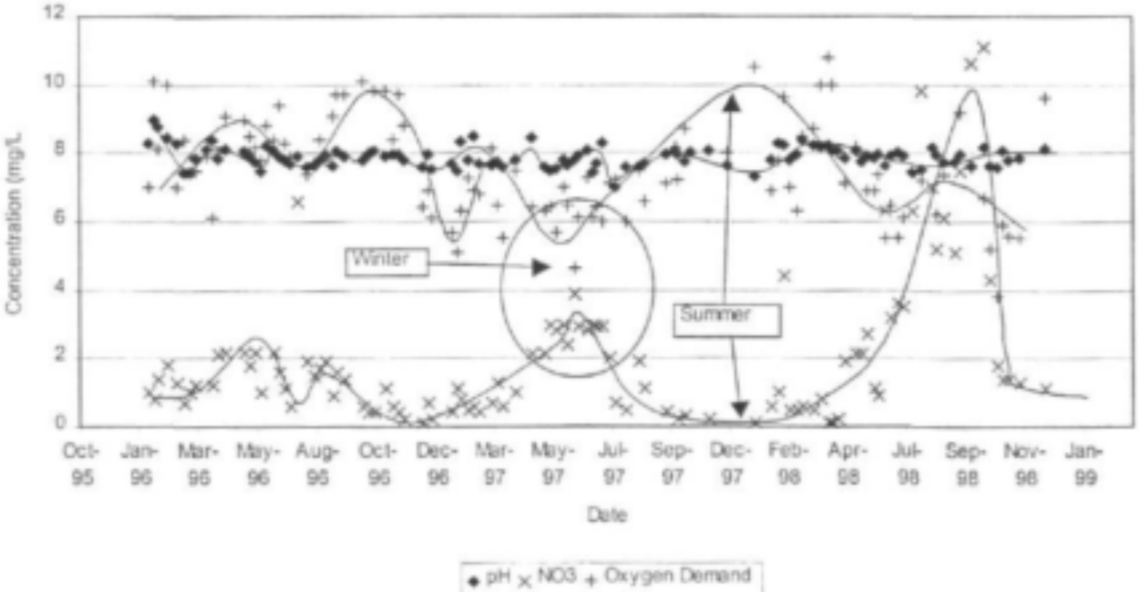

Figure 15 Polokwane WWTW discharge water quality (pH, Nitrate and Biochemical Oxygen Demand)

5.2.3 Groundwater Quality

Two aquifer systems are recognised within the study area; the phreatic alluvium and the semi-confined granitic gneiss aquifers. The following section gives an overview of the water quality in these two aquifer systems.

An extensive groundwater exploration programme was conducted within the Sand River study area by the DWAF (Orpen, 1986). The following water quality trends were observed:

- The electrical conductivities of groundwater in most cases decreased with depth.
- The quality of groundwater tended to improve with pumping (observed during the test-pumping).
- Water quality appeared to improve from south to north.

It should however be noted that all the DWAF (Orpen, 1986) boreholes were drilled down-stream of the Waste Water Treatment Works. This may partly explain the observed improvement in water quality from south to north during the drilling programme. Orpen (1986) describes the groundwater sampled during the drilling programme as a Na-HCO_3 water with high levels of chloride, and its quality only marginally suitable for domestic use.

The following table (Table 5) represents data collected during exploration drilling (Orpen, 1986), and illustrates the characteristics of water quality in the sand and gneiss aquifers. Also included is a chemical analysis typical of the water quality observed in boreholes drilled in the Polokwane town area (Du Toit, 1986).

Table 5 Comparison of water quality observed in the alluvial and granite gneiss aquifers.

	Sand	*Granite-gneiss (Study Area)*		*Granite-gneiss (Town)*	
	G-36269	G-36229	G-36271	G-35860	G-36239
Potassium as K mg/L	3	5	6	6.8	11
Sodium as Na mg/L	127	125	84	30	85
Calcium as Ca mg/L	29	32	49	16	32
Magnesium as Mg mg/L	39	38	40	19	46
Ammonia as N mg/L	0.04	0.03	0.03	0.04	0.05
Sulphate as SO4 mg/L	32	20	35	21	11
Chloride as Cl mg/L	107	101	108	14	62
Alkalinity as CaCO3 mg/L	317	330	237	126	339
Nitrate plus nitrite as N mg/L	0.8	0.1	8.2	4.23	2.01
Conductivity mS/m @25°C	97	95.7	92.4	37.1	88.5
pH (Lab)	7.9	8.1	7.6	7.5	7.9
Total Dissolved Solids (Calc)	745	747	680	301	701
Hardness as CaCO3 mg/L	233	236	287	118	269
HCO_3	386	402	289	154	434
Silica as Si mg/L	16	22	33	21.9	29.4
Fluoride as F mg/L	0.5	0.5	0.4	0.3	0.9
Phosphate as P mg/L	0	0.06	0	0.02	0.02

Of note is the similarity in the element concentrations observed in the two aquifer systems in the Sand River area. The close similarities in the chemical composition of the water point to a hydraulic connection between the two aquifers. The data should however be interpreted with caution as the construction of these boreholes are such that samples collected from the two deeper boreholes are a composite of the water found in both the gneiss and sand aquifers. The two boreholes that represent samples taken from the larger town area show large variation. The location of these boreholes in an urban setting make them susceptible to pollution (Du Toit, 1986). The quality in these boreholes do however give an indication of natural background water quality in the Granitic-gneiss aquifer.

Proper interpretation of groundwater quality required the construction of boreholes that would target either the alluvium or the gneiss aquifer for sampling (section 3.2.1). The chemical analyses obtained from the newly constructed boreholes (S1B1 - gneiss and S1B6 - alluvium) together with the average WWTW discharge quality and analysis results from two boreholes that penetrate the gneiss aquifer 5 km up-stream of the WWTW are presented in Table 6.

The analysis shows that water quality of the gneisses up-stream of the WWTW differ significantly from down-stream of the WWTW. This applies particularly to the concentration of sodium, sulphate, chloride, nitrate, and total hardness. The data further shows that the quality of water in the alluvium and gneiss down-stream of the WWTW are remarkably similar. This further supports the contention that a hydraulic connection exists between the alluvium and the gneiss aquifers.

Sulphate concentrations comparable with those in the alluvium aquifer are observed in the gneiss borehole S1B1. This points to the flow of water from the alluvium aquifer to the gneiss aquifer. The salt concentrations (electrical conductivity and total dissolved solids) in the *alluvium aquifer and the gneiss aquifer at the end of the constant discharge test* is also much higher than in the samples representative of conditions up stream of the WWTW. This is likely the result of the influence of discharge water from the WWTW (see section 5.5).

Up-stream of the WWTW the similarity in quality of the surface water and the gneiss boreholes probably indicate that water in the gneiss aquifer is, derived from direct surface recharge. The only slight differences in element concentrations are those for calcium and magnesium. The difference is small and can be explained through dissolution of the minerals during infiltration. Of interest is the higher sulphate concentration in the Sand River samples compared to samples of the gneiss water. This may be a result of human impact further up stream in the Sand River. Activities up-stream of the sampling point includes small holding agriculture, light industries and human settlements.

Table 6 Comparison of water quality observed in the gneiss aquifer, the alluvium aquifer, and the Sand River.

	Up-stream of WWTW			*Down-stream of WWTW*		
Borehole No:		**Sterk-Bh1**	**Town (G35860)**	**WWTW**	**SIB6**	**SIB1**
Lithology:	Sand River	Gneiss	Gneiss	Discharge	Alluvium	Gneiss
Sample date:	Average 1995-'99	1997	1986	Average 1996-'98	18/6/2000	19/6/2000
Potassium as K mg/L	5.6	5.1	6.8	-	8.8	3.6
Sodium as Na mg/L	31.4	31	30	151.9	148	129
Calcium as Ca mg/L	23.5	34	16	-	50	32
Magnesium as Mg mg/L	14.5	27	19	-	25	35
Ammonia as N mg/L	0.075	-	0.04	6.1	0.83	<0.1
Sulphate as SO4 mg/L	23	7	21	46	44	41
Chloride as Cl mg/L	30	26	14	150	123	130
Alkalinity as CaCO3 mg/L	130	215	126	237	333	282
Nitrate plus nitrite as N mg/L	0.18	2.7	4.23	2.7	<0.1	0.1
Dissolved Organic Carbon		-	-		2.9	1.4
Conductivity mS/m @25°C	40	47	37	104	108	100
pH (Lab)	8.0	8.1	7.5	7.9	8.4	7.9
Saturation pH (pHs) (20°C)	-	-	-		7.4	7.7
Total Dissolved Solids (Calc)	289	301	301	632	691	640
Hardness as CaCO3 mg/L	-	85	118	-	227	226
Sodium Absorption Ratio	-	-	-	-	4.3	3.7

5.2.4 Recharge assessment based on Groundwater quality

Historical chloride values were used to assess the volume of WWTW water that infiltrate the gneiss aquifer. The chloride value of water from the gneiss aquifer up-stream of the WWTW is between 14 and 26 mg/L (see Table 6). Water samples obtained from the gneisses located in the area surrounding Polokwane yielded values of 14, 19, 23, 26, 36 and 57 mg/L (Du Toit, 1986, and pers. comm. J. Weaver).Water quality in the Polokwane Town area shows a large variation, which is ascribed to urban pollution point sources. Orpen (1986) also reports a wide range of water qualities in the wider area that surrounds Polokwane. Here Chloride values vary between 10 and 250 mg/L. For the purposes a realistic average natural chloride content for the gneisses in the study area of 29 mg/L is assumed. The chloride content down stream of the WWTW is 130 mg/L, while the chloride content in the WWTW discharge water is 150 mg/L. From this the recharge contribution from the WWTW is calculated as follows:

Recharge from WWTW = (130 - 29)/150 x 100 = 67%

Considering that discharge from the WWTW has gone on for a number of years, this percentage does not necessarily reflect current infiltration. Should this figure be taken to represent annual infiltration, an annual infiltration of approximately 4.3. M m^3/a recharge water is derived from the WWTW. This compares reasonably well with the value of 5.2 M m^3/a derived from the pumping test data.

5.3 Seasonal Variations in Water Quality

Although some long-term data on groundwater quality is available from boreholes along the Sand River, the frequency with which water samples were taken and analysed is limited. None of the available data precedes the period at which WWTW discharge commenced.

5.3.1 Hardness and Alkalinity

The hardness and alkalinity of river water is generally less than that of the natural groundwater during most of the year. The hardness and alkalinity of the groundwater from an aquifer that obtains a substantial part of its recharge from the river increases gradually during long periods of low flow, and decrease rapidly during periods of flood. The climate of the Polokwane area is dominated by periods of high rainfall in the summer months and periods of very little precipitation during winter (section 2.1). Flow in the Sand River mirror this, with the period of highest flow during the summer months.

5.3.2 General comments

Interpretation of the borehole chemistry data indicate a slight to moderate rise in hardness values in winter compared to summer for many of the boreholes located in the river alluvium. Boreholes 11, 12 and 13 are the exception, and show a slight decrease in hardness during the winter months (Appendix 3). These boreholes are constructed to abstract water from both the river alluvium and the gneisses. Therefor samples represent a composite of the two aquifers.

5.4 *Quality variation with distance from the WWTW*

Graphs showing variation in water quality with distance from the WWTW are shown in Appendix 4. When looking at the concentrations of the major elements with distance from the WWTW in direction of groundwater flow we generally see very little change in the element concentrations. If anything the data shows that water quality improves with distance from the WWTW discharge point. The trend seems to be for the TDS concentration to start high with a gentle decline down the flow path. This improvement in quality may partly be ascribed to dilution. The exception to this trend is borehole 2, which generally shows lower concentrations than its two closest boreholes. In the case of the sampling and analysis done in October 1996 a drop in TDS concentration of about 400 mg/L was observed between boreholes 1 and 3, with a rise in TDS concentration of about 300 mg/L between boreholes 3 and 4.

5.5 *Water quality with depth*

A hydrochemical profile with depth was done on boreholes S1B1, S1B3 and S1B6. Figures 16 and 17 show the observed change in electrical conductivity (EC) and pH with depth for each of these boreholes. These profiles do not however provide sufficient information to allow for the quantification of the hydraulic linkages between the three units. The depth profiles do allow for the following observations to be made:

1. The electrical conductivity of the alluvium water (S1B6) is higher than that of the dolerite (S1B3) or gneiss waters (S1B1);

2. The electrical conductivities of the gneiss water within the cased portion of the borehole (3 - 23 m) is not representative of the gneisses;

3. The electrical conductivities and pH of the dolerite water is very similar to the gneiss water;

4. The pH of the gneiss (S1B1) and dolerite (S1B3) waters were influenced by the cement used to case off the upper parts of these boreholes.

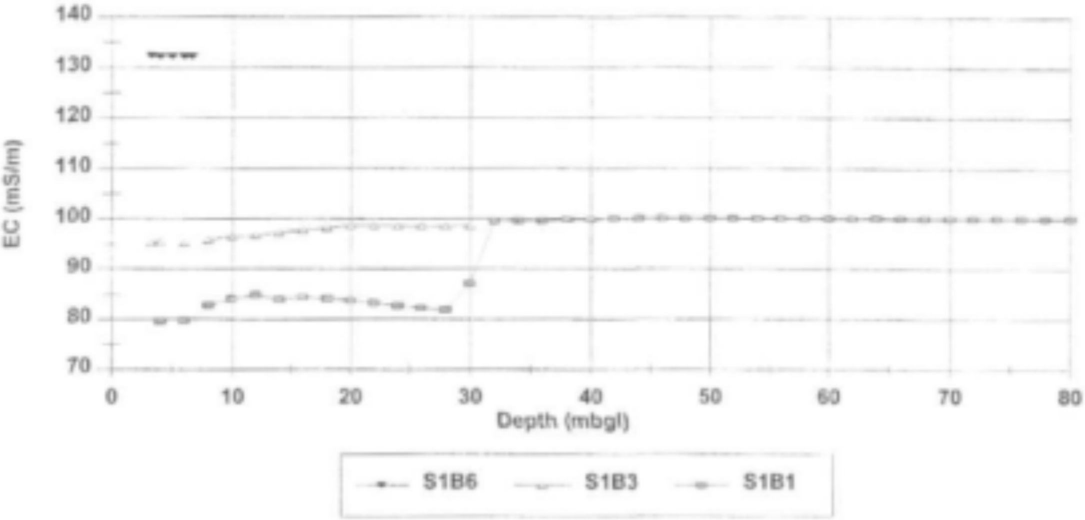

Figure 16 Electrical conductivity profile with depth in boreholes S1B1, S1B3 and S1B6

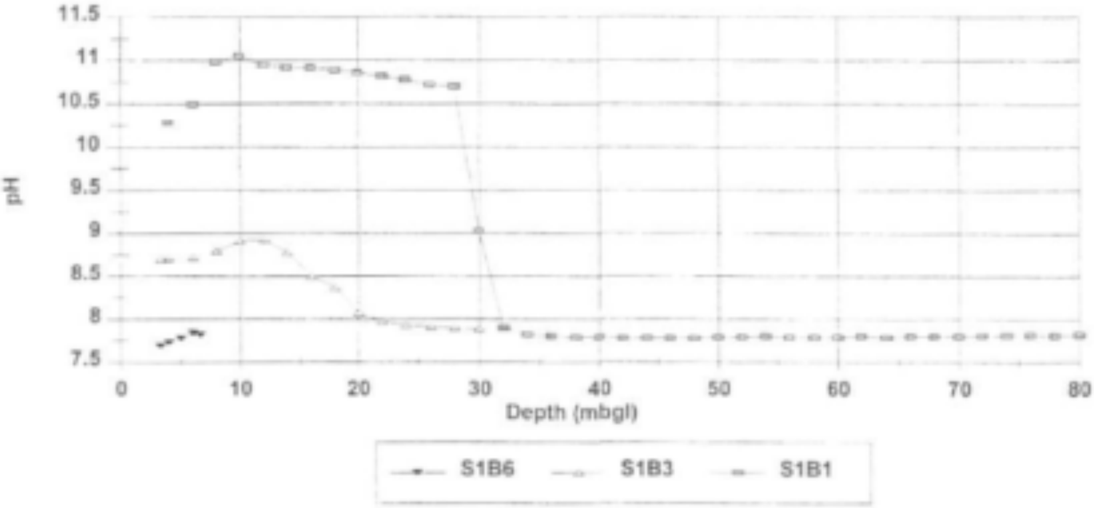

Figure 17 pH profile with depth in boreholes S1B1, S1B3 and S1B6

6. CONCLUSIONS

The quality of water abstracted from boreholes in the gneisses located above the WWTW discharge is generally good. Of note is that this water differs in quality from the groundwater abstracted downstream of the WWTW. This difference in quality is ascribed to the impact of the water discharge from the WWTW. It is further observed that a close similarity in quality exists between water abstracted from the alluvium aquifer and the gneiss aquifer below the WWTW. This supports the contention that a hydraulic connection exists between the alluvium aquifer and the gneiss aquifer, and that recharge of the gneiss aquifer by discharge water from the Polokwane WWTW is taking place.

From the available major anions/cations data, it appears if the quality of the water that is discharged from the Polokwane WWTW does not contain chemical elements in concentrations that pose serious risks to consumers. The high nitrate concentrations of the WWTW water during certain periods of the year does pose a health threat to sensitive groups like infants. This and the relatively high dissolved solid content of the groundwater can be managed by blending it with the town's surface water sources.

It is not known which by-products are produced during the waste water treatment process. Such by-products may include trihalomethanes and other organic compounds. Should such compounds enter the domestic water supply they would represent a health threat to water users.

The process of wastewater infiltration through the alluvium along the Sand river allows for the reuse of Polokwane's waste water. Such reuse of water could be of economic and social benefit in an area that regularly experiences water shortages. Each year about 6 million m^3 of water is released from the Polokwane Waste Water Treatment Works. The results of the chloride assessment and the pumping tests suggest that about 4 to 5 Mm^3/a can infiltrate from the alluvium to the gneisses. Although there are drawbacks to these analysis, they nevertheless indicate that a high proportion of the WWTW's discharge can infiltrate into the hardrock aquifer from which Polokwane gets its groundwater. Rough water balance calculations indicate that ~ 3.4 - 4.1 Mm^3, or 58 - 70 % of water discharged from the WWTW is available for recharging the gneissic aquifer.

Additional work could be done in order to optimise the functioning of recharge scheme. Of particular concern in the present design is the lack of an unsaturated zone to assist in the purification of the recharge water. Filtration through an unsaturated zone could address concerns about the pollution of the groundwater by microbial contaminants. In the present system it is possible that some bacteria, viruses or parasites could survive the bank infiltration process and contaminate the groundwater. The survival rate of such organisms can be expected to be low in a system that requires filtration through an unsaturated zone, while the chlorination of domestic water supplies should result in their total die-off.

7. RECOMMENDATIONS

It is advised that the Polokwane municipality implement the following as a means to optimise the present groundwater recharge process.

1. Assess the value of increasing infiltration of natural runoff by creating space within the aquifer during summer months. This could be done by abstracting groundwater at high rates during this period. Assuming that the natural rainfall runoff is of good quality this could lead to improving the quality of the gneiss water.
2. Modify the groundwater quality assessment programme so that regular and consistent analysis are done, and include determinants such as potassium and organics like trihalomethane.
3. The occurrence of organics (like trihalomethanes) in selected production borehole should be checked at regular intervals. Testing once a year should be sufficient to meet present requirements.
4. Analysis of phosphorus, total nitrogen and ammoniacal nitrogen in addition to oxidised nitrogen would provide a more complete picture of nutrient discharge and likely processes in the waste water discharge.
5. Implement a groundwater level monitoring programme which include boreholes adjacent and away from the river. Ideally this should be done before and after the rainy season.
6. Assess the groundwater quality below the WWTW for organics. If it is found that this is a concern, then further pre-treat the waste water.
7. The pollution sources that could impact the water quality of the wellfield should be more strictly managed. Of particular concern is sources of nitrate pollution such as concentrated cattle farming.
8. Water quality samples could be obtain with greater ease and accuracy from the Sand River production boreholes, through the installation of sampling taps at each well head.
9. Future studies could attempt to calculate recharge through assessing the water balance of the Sand River downstream of the WWTW. In this method recharge would be estimated differently depending on whether the stream is flowing or not.

   A. If there is no flow; estimate the wetted area over which evapotranspiration takes place, and take a factor of between 0.5 and 0.7 of pan evaporation to give the rate of evapotranspiration, which can be deducted from the wastewater discharge rate to give a net recharge.
   B. If there is flow; the increase in stream head and wetted area should be determined, and used with an analytical model to calculate the incremental recharge attributable to wastewater discharge.

It is possible to use the permeability or leakage rate derived from (A) to parameterise the model in (B).

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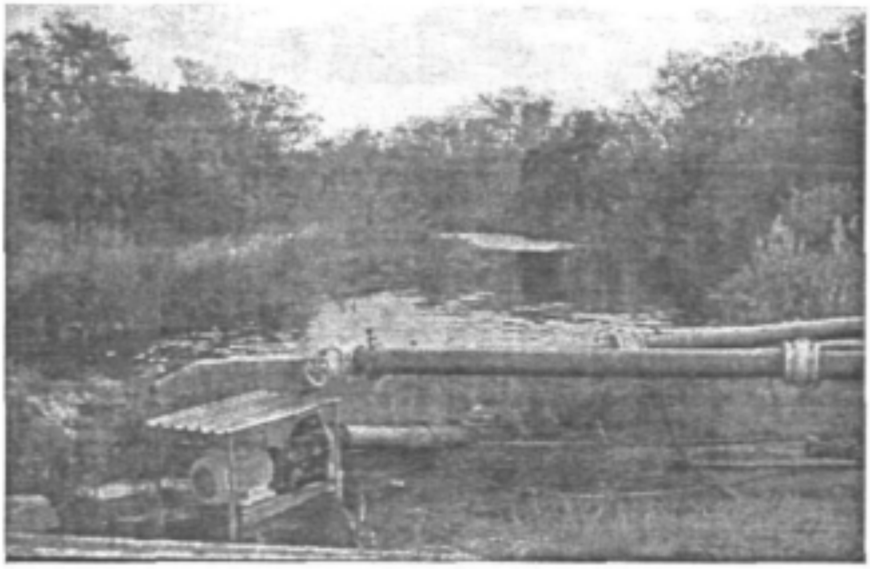
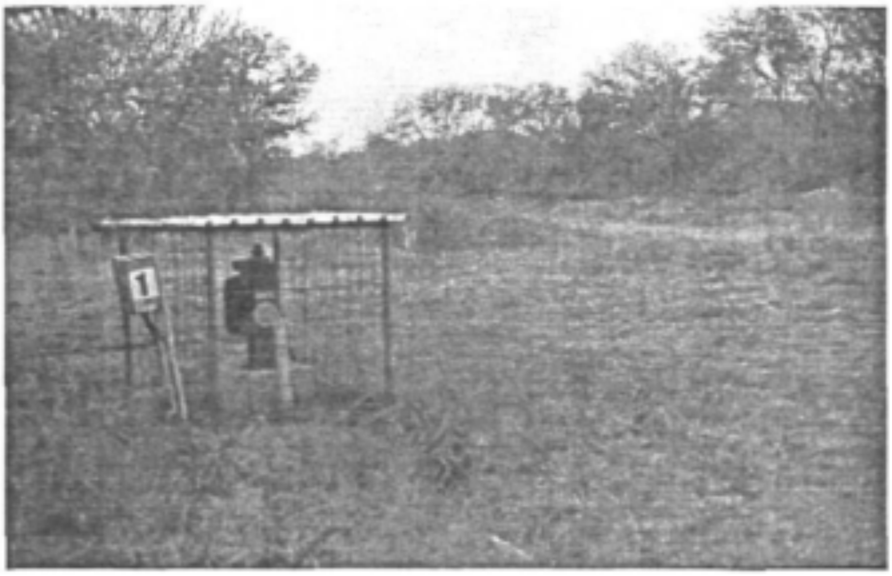
Plates and Description

Plate 1 Surface Water Abstraction from the Sand River



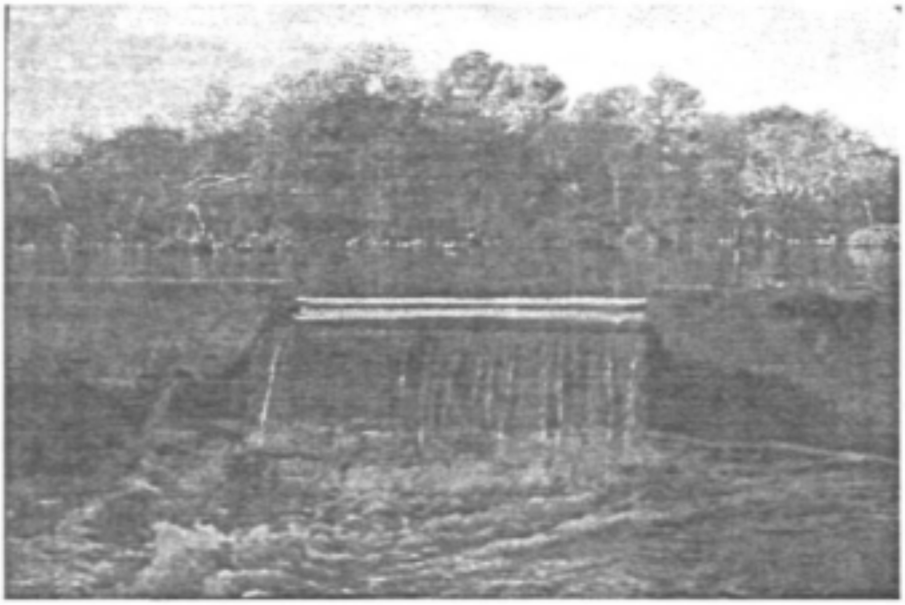
Farmers abstract water directly from the Sand River stream channel. Up-stream of the WWTW there is almost no flow in the river during the dry winter months. Downstream of the WWTW flow during winter is almost completely the result of discharge from the treatment works (see plate 3).

Plate 2 A Groundwater Abstraction Borehole along the Sand River



A number of abstraction boreholes like this one are located along the Sand River. These boreholes abstract water for municipal supply during periods of water stress. During such periods groundwater may contribute close to 5 million m^3 to the towns water supply. The average supply for the period 1976 to 1985 was 3.08 million m^3. In addition to the groundwater being pumped for the town supply, farms along the banks of the Sand and Blood Rivers also abstract large quantities of water for their agricultural purposes.

Plate 3 Polokwane Waste Water Treatment Works Oxidation Pond 3 and discharge to the Sand River



Following primary treatment waste water is directed to open surface oxidation ponds. The water flows through 3 ponds, before discharging to the Sand River (shown here). Retention time in the 3 oxidation ponds is about 2 weeks. Each year about 6 million m^3 of water is discharged to the Sand River. For most of the year this discharge water is the biggest and sometimes only contribution to flow in the Sand River (See plate 1).

Plate 4 Sand Mining along the Sand River

Sand thickness along the Sand River reaches a depth of upto 30 meters. Some mining takes place along the Sand River to exploit some of this high quality building sand. Present mining activities takes place mostly in the north of the study area, near the confluence of the Sand and Blood Rivers. Should this mining activity continue, it may well influence the storage capacity of the alluvial aquifer in the long term.

APPENDIX 4.1:

Borehole logs of newly drilled boreholes

WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

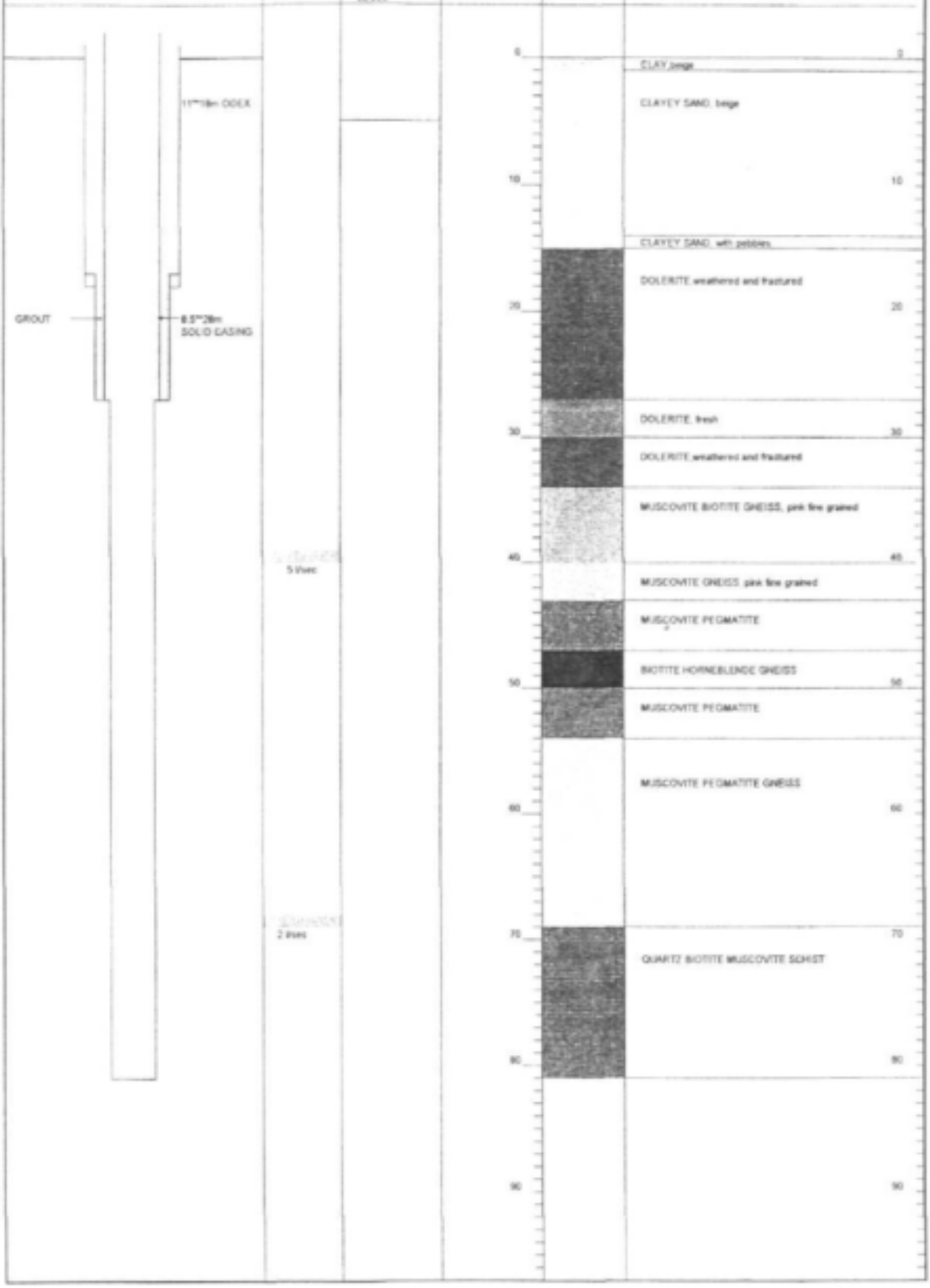
TEL NO (015) 291-5181
FAX NO (015) 291-5268

PERCUSSION DRILLING RECORD

CLIENT : WRC
PROJECT : SANDRIVER AQUIFER
SITE : SANDRIVER
LOG : P A WILKEN
DRILLER : DWAF
DATE STARTED :

JOB № : 99025
BOREHOLE № : S181
WATER TABLE :
ELEVATION :
LATITUDE :
LONGITUDE :

BOREHOLE CONSTRUCTION | WATER STRIKE | STATIC WATER LEVEL | DEPTH (METERS) | PROFILE | MATERIAL DESCRIPTION



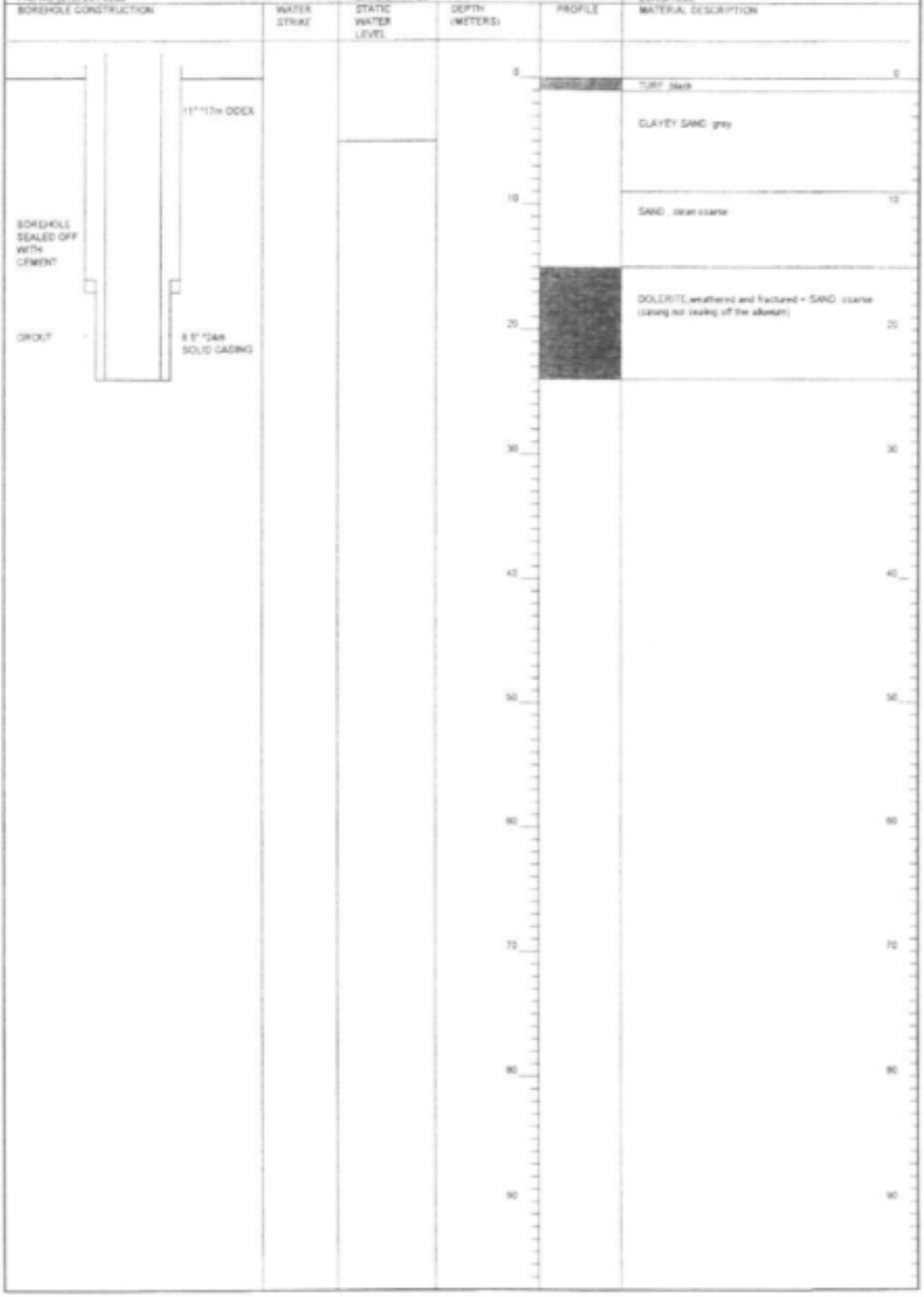
WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

TEL NO (015) 291 5191
FAX NO (015) 291 5268

PERCUSSION DRILLING RECORD

CLIENT : WRC
PROJECT : SANDRIVER AQUIFER
SITE : SANDRIVER
LOG : P A WILKEN
DRILLER : DWAF
DATE STARTED :

JOB № 99025
BOREHOLE № S182
WATER TABLE
ELEVATION
LATITUDE
LONGITUDE



TURF, black

CLAYEY SAND, grey

SAND, clean coarse

DOLERITE,weathered and fractured + SAND, coarse
(casing not sealing off the alluvium)

WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

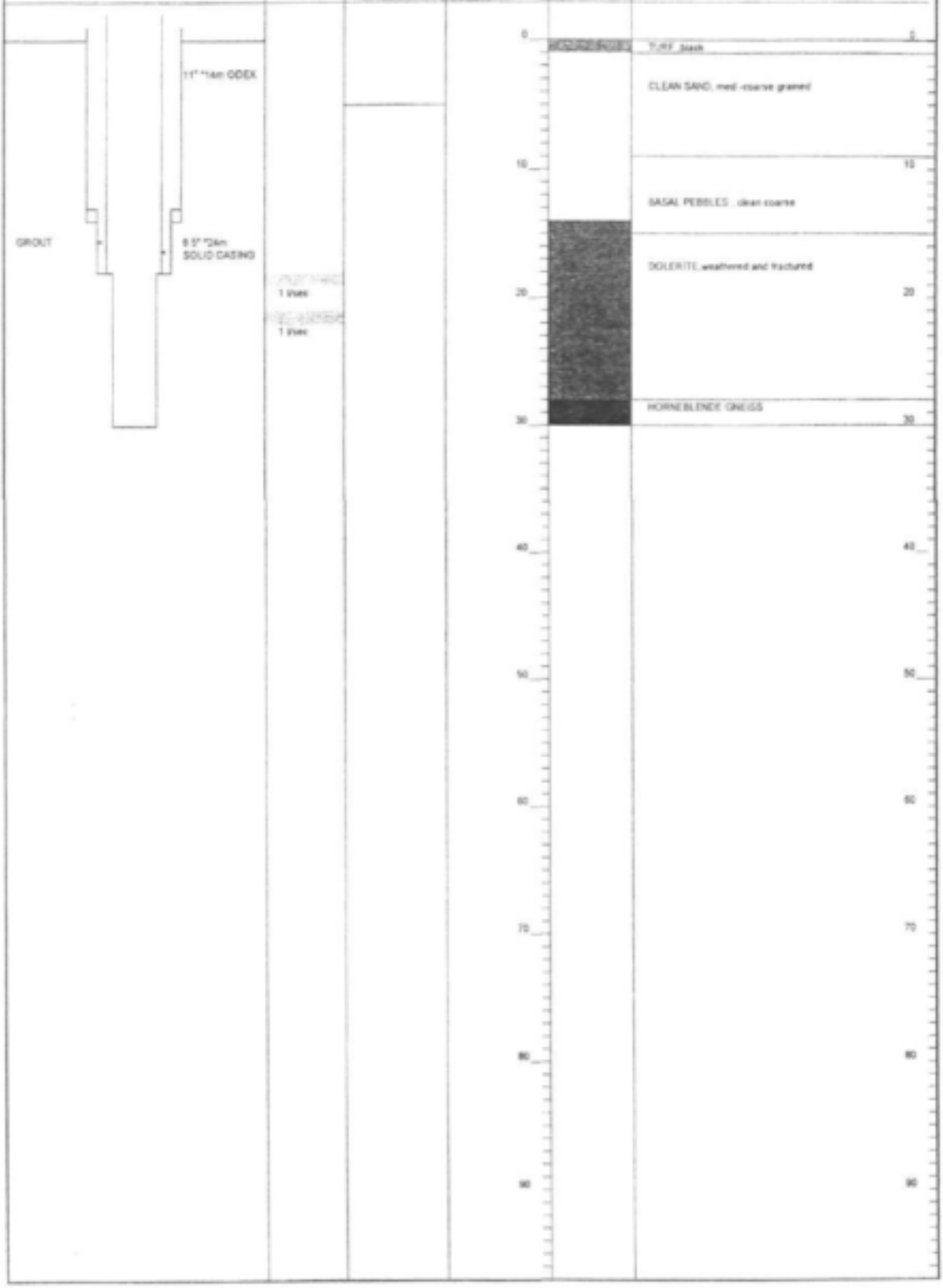
TEL NO (015) 291-5191
FAX NO (015) 291-5298

PERCUSSION DRILLING RECORD

CLIENT : WRC
PROJECT : SANDRIVER AQUIFER
SITE : SANDRIVER
LOG : P A WILKEN
DRILLER : DWAF
DATE STARTED :

JOB № : 99025
BOREHOLE № : S1B3
WATER TABLE :
ELEVATION :
LATITUDE :
LONGITUDE :

BOREHOLE CONSTRUCTION | WATER STRIKE | STATIC WATER LEVEL | DEPTH (METERS) | PROFILE | MATERIAL DESCRIPTION



TURF, black

CLEAN SAND, med.-coarse grained

BASAL PEBBLES , clean coarse

DOLERITE, weathered and fractured

HORNEBLENDE GNEISS

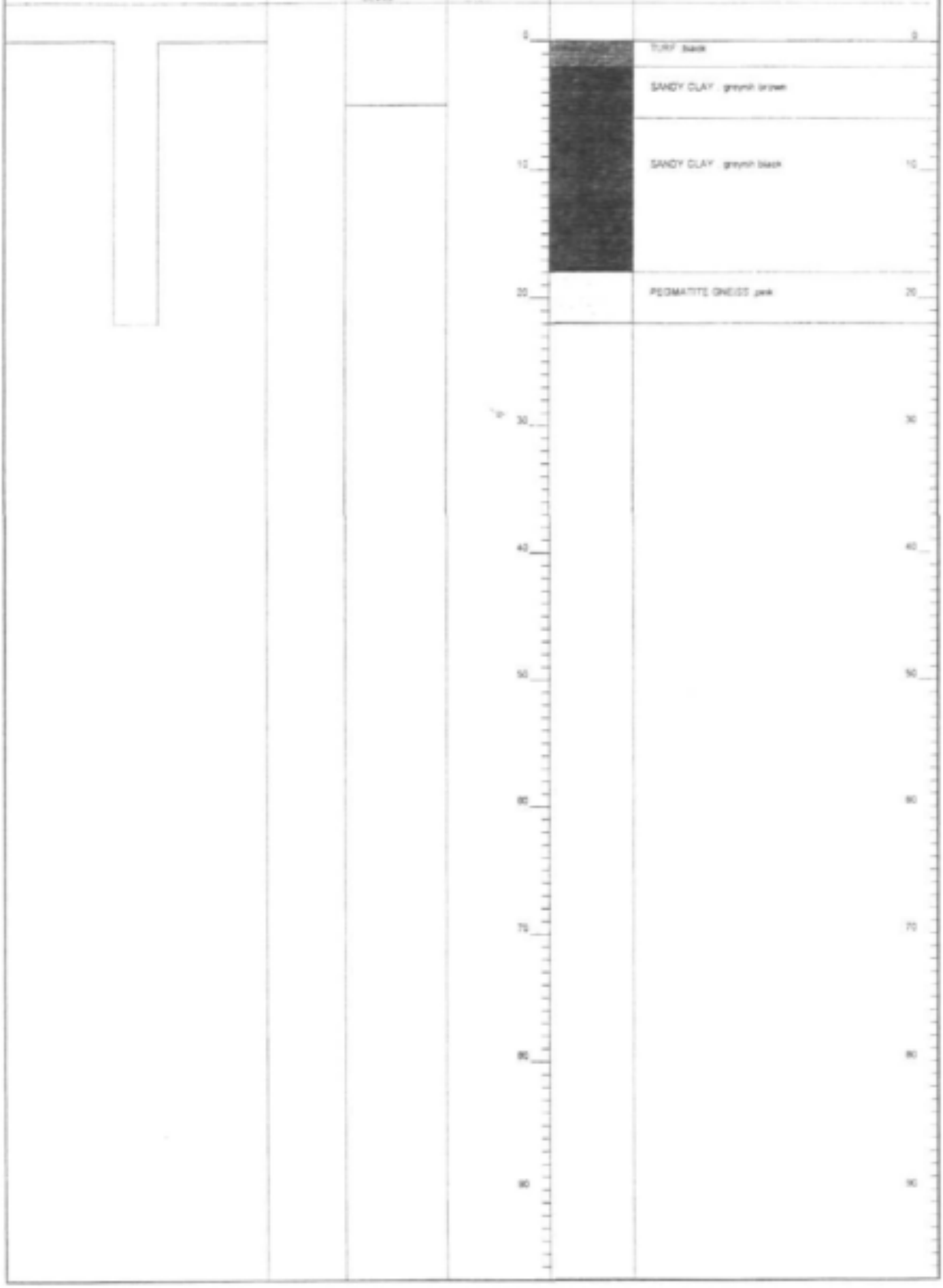
WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

TEL NO (015) 291-5191
FAX NO (015) 291-5268

PERCUSSION DRILLING RECORD

CLIENT	WRC	JOB №	99025
PROJECT	SANDRIVER AQUIFER	BOREHOLE №	S184
SITE	SANDRIVER	WATER TABLE	
LOG	P A WILKEN	ELEVATION	
DRILLER	DWAF	LATITUDE	
DATE STARTED		LONGITUDE	

BOREHOLE CONSTRUCTION | WATER STRIKE | STATIC WATER LEVEL | DEPTH (METERS) | PROFILE | MATERIAL DESCRIPTION



TURF . black

SANDY CLAY . greyish brown

SANDY CLAY . greyish black

PEGMATITE GNEISS . pink

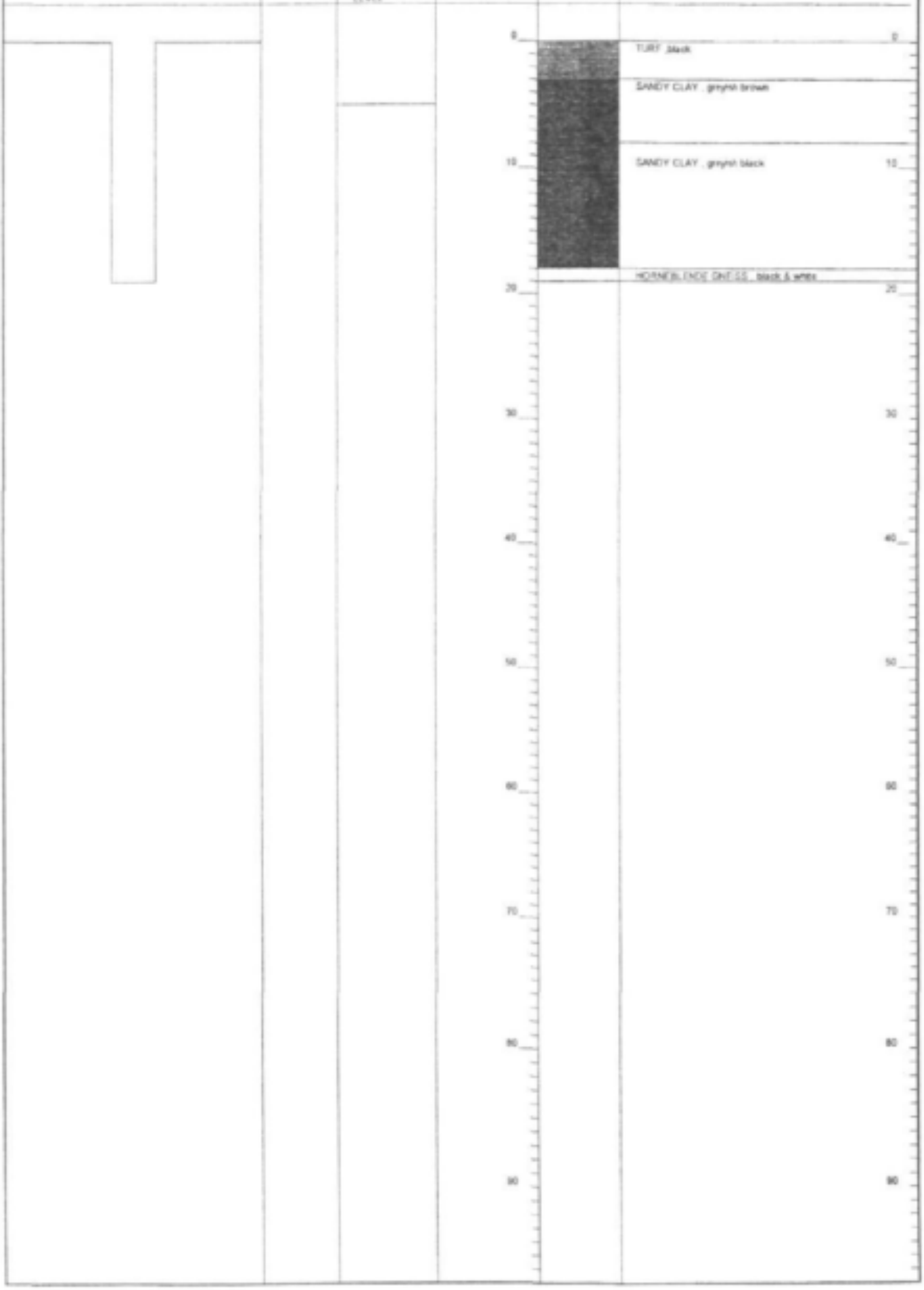
WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

TEL NO (015) 291-5191
FAX NO (015) 291-5768

PERCUSSION DRILLING RECORD

CLIENT :	WRC	JOB Nr :	99025
PROJECT :	SANDRIVER AQUIFER	BOREHOLE Nr :	S185
SITE :	SANDRIVER	WATER TABLE :	
LOG :	P A WILKEN	ELEVATION :	
DRILLER :	DWAF	LATITUDE :	
DATE STARTED :		LONGITUDE :	

BOREHOLE CONSTRUCTION	WATER STRIKE	STATIC WATER LEVEL	DEPTH (METERS)	PROFILE	MATERIAL DESCRIPTION



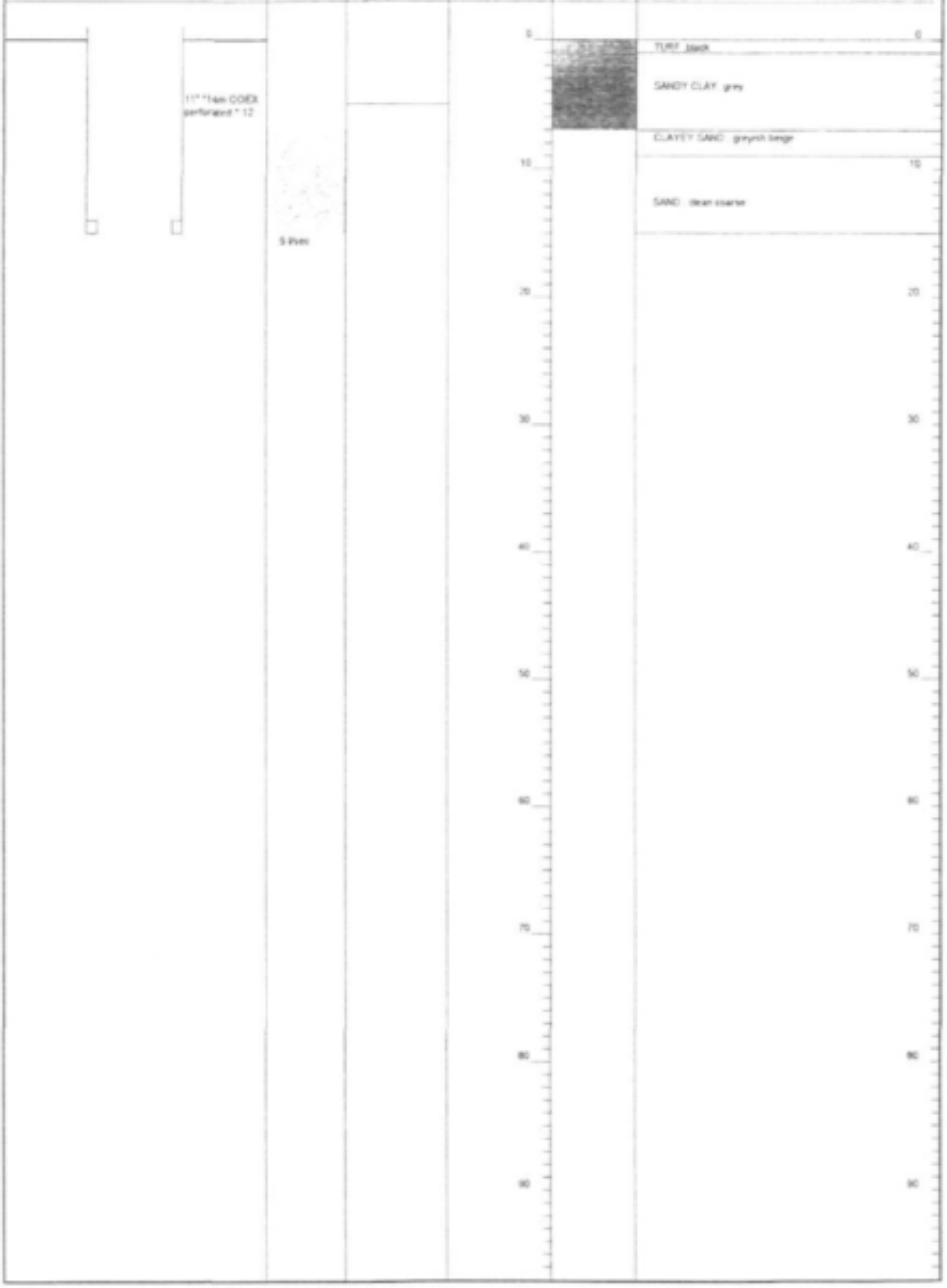
WATER SYSTEMS MANAGEMENT cc
P O BOX 3356
PIETERSBURG
0700

TEL NO (015) 291-5191
FAX NO (015) 291-5288

PERCUSSION DRILLING RECORD

CLIENT	WRC	JOB №	99025
PROJECT	SANDRIVER AQUIFER	BOREHOLE №	S186
SITE	SANDRIVER	WATER TABLE	
LOG	P A WILKEN	ELEVATION	
DRILLER	DWAF	LATITUDE	
DATE STARTED		LONGITUDE	

BOREHOLE CONSTRUCTION	WATER STRIKE	STATIC WATER LEVEL	DEPTH (METERS)	PROFILE	MATERIAL DESCRIPTION



TURF black

SANDY CLAY grey

CLAYEY SAND greyish beige

SAND clean coarse

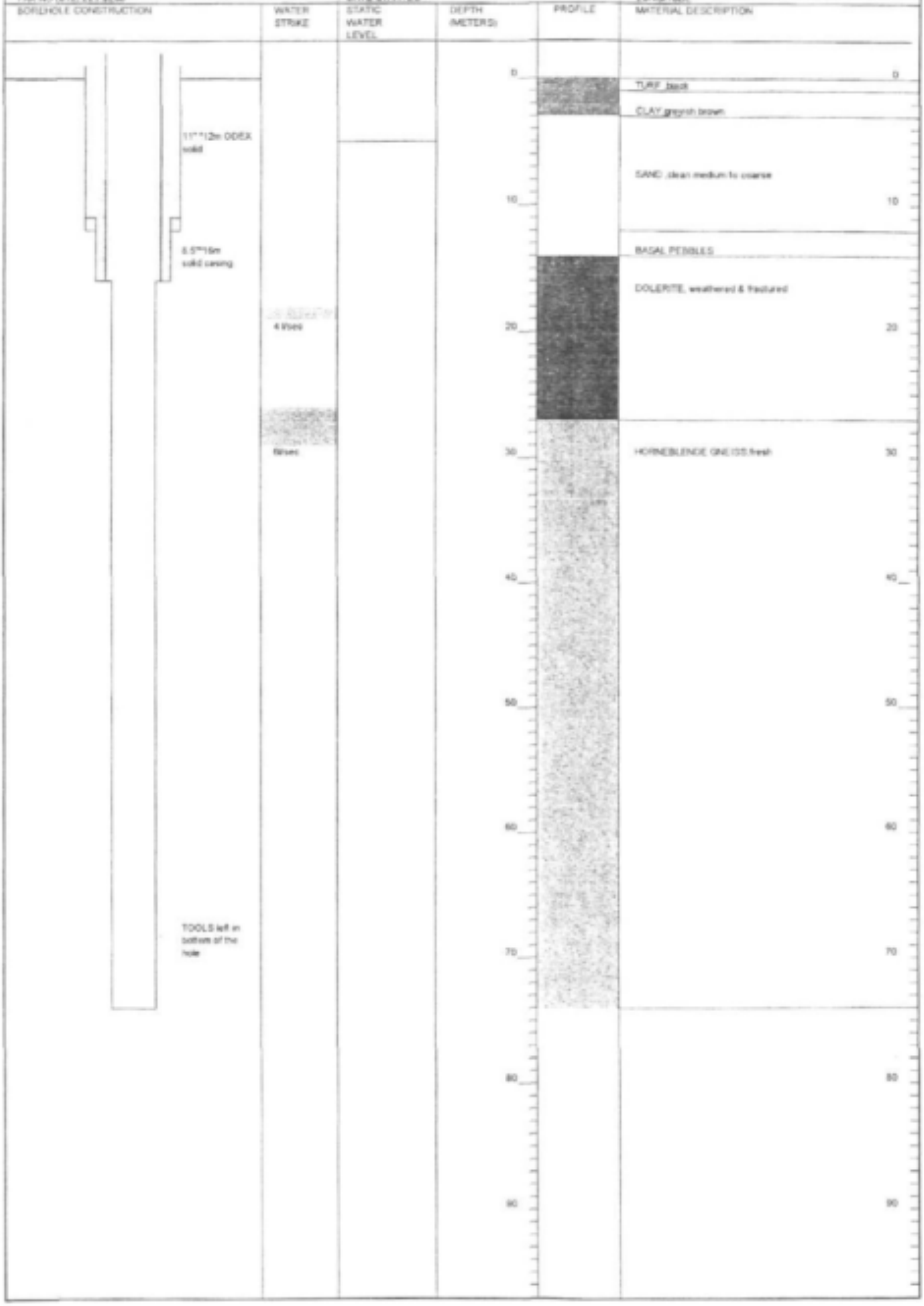
WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

TEL NO (015) 291-5191
FAX NO (015) 291-5268

PERCUSSION DRILLING RECORD

CLIENT : WRC
PROJECT : SANDRIVER AQUIFER
SITE : 2 SANDRIVER
LOG : P A WILKEN
DRILLER : DWAF
DATE STARTED :

JOB Nr : WK025
BOREHOLE Nr : S2B1
WATER TABLE : 5 mbcollar
ELEVATION :
LATITUDE :
LONGITUDE :



WATER SYSTEMS MANAGEMENT cc
P O BOX 3316
PIETERSBURG
0700

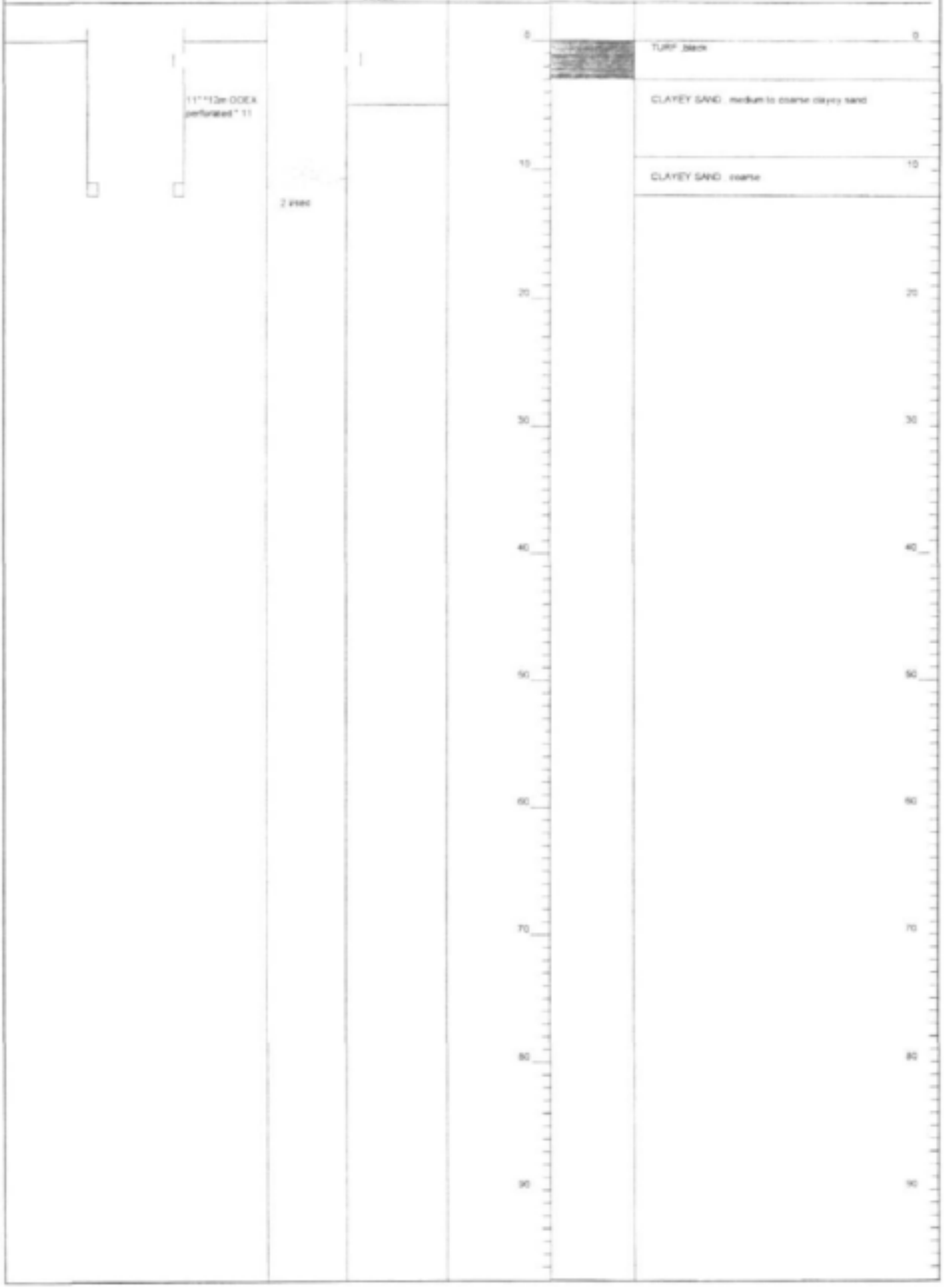
TEL NO (015) 291-6191
FAX NO (015) 291-5266

PERCUSSION DRILLING RECORD

CLIENT : WRC
PROJECT : SANDRIVER AQUIFER
SITE : SANDRIVER
LOG : P A WILKEN
DRILLER : DWAF
DATE STARTED :

JOB Nr : 99025
BOREHOLE Nr : S282
WATER TABLE :
ELEVATION :
LATITUDE :
LONGITUDE :

BOREHOLE CONSTRUCTION	WATER STRIKE	STATIC WATER LEVEL	DEPTH (METERS)	PROFILE	MATERIAL DESCRIPTION



APPENDIX 4.2:

Pumptest Drawdown Curves

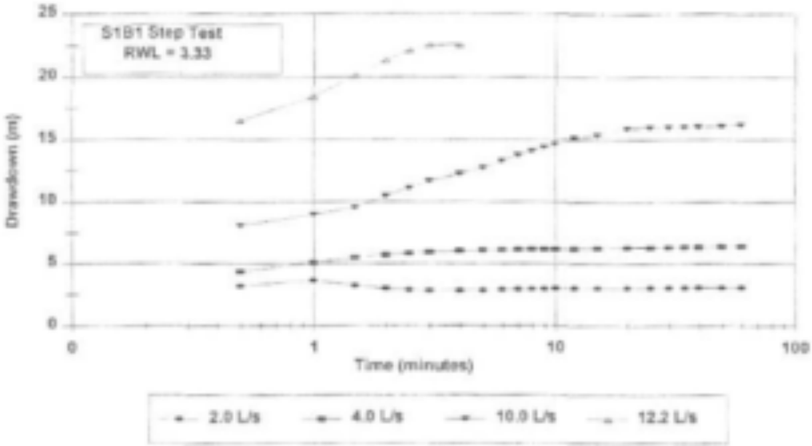


Figure 1 Pumping borehole S1B1 (unweathered gneiss) during step test

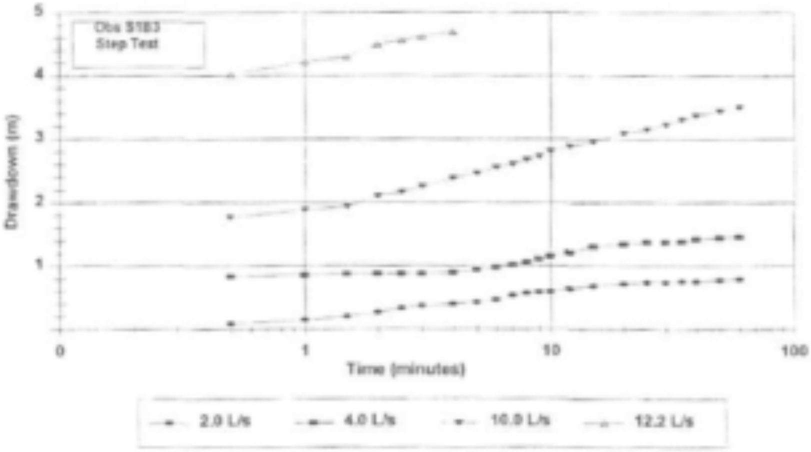


Figure 2 Observation borehole S1B3 (weathered gneiss), pumping borehole S1B1 (step test)

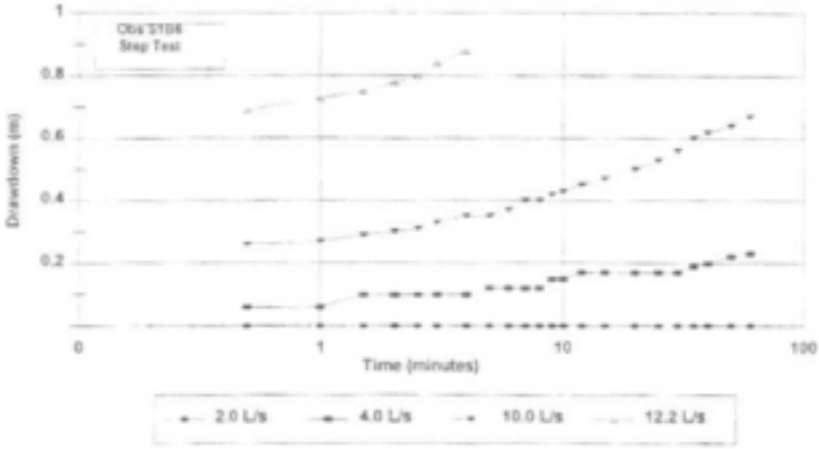


Figure 3 Observation borehole S1B6 (alluvium), pumping borehole S1B1 (step test)

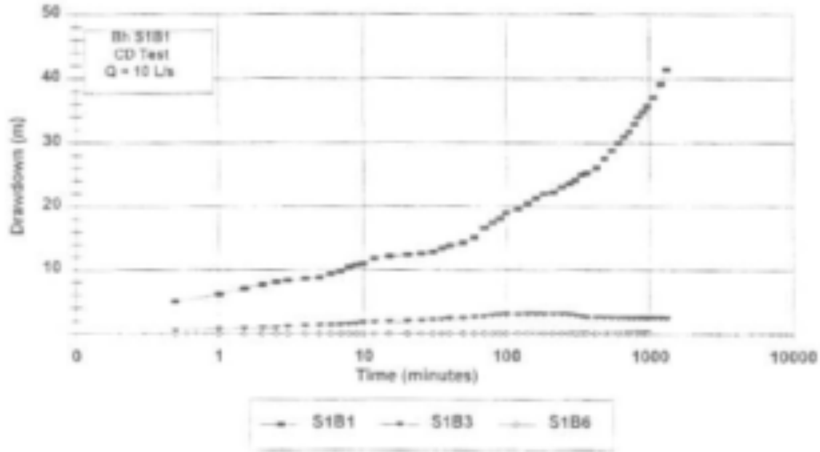


Figure 4 Borehole S1B1 (unweathered gneiss), during constant discharge test

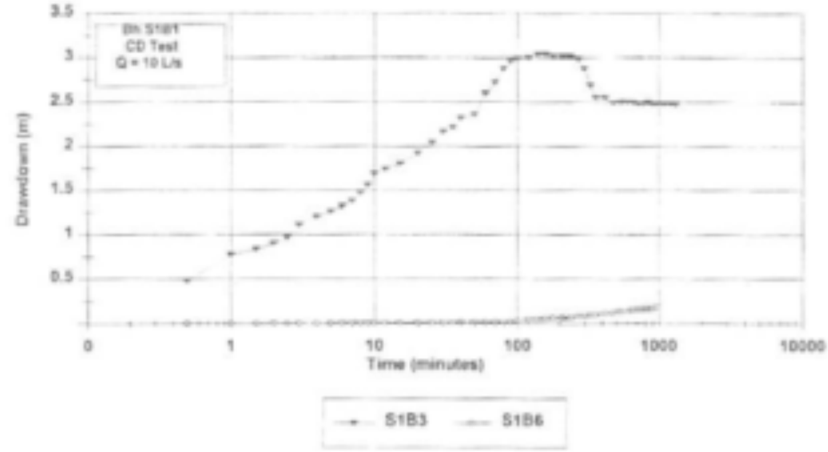


Figure 5 Observation boreholes S1B3 and S1B6, pumping borehole S1B1 (constant discharge test)

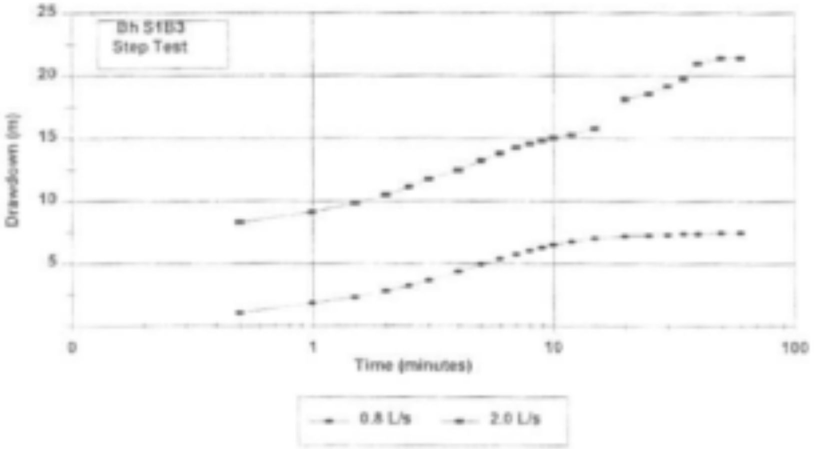


Figure 6 Pumping borehole S1B3 during step test

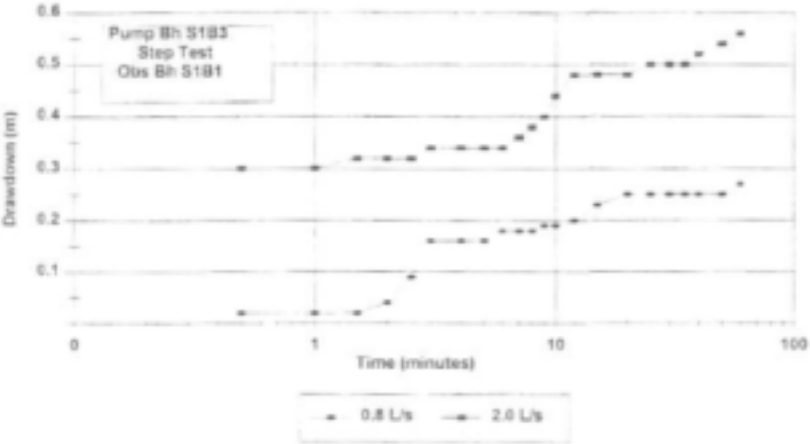


Figure 7 Observation borehole S1B1 (unweathered gneiss), pumping borehole S1B3 (wethered gneiss) (during the step test)

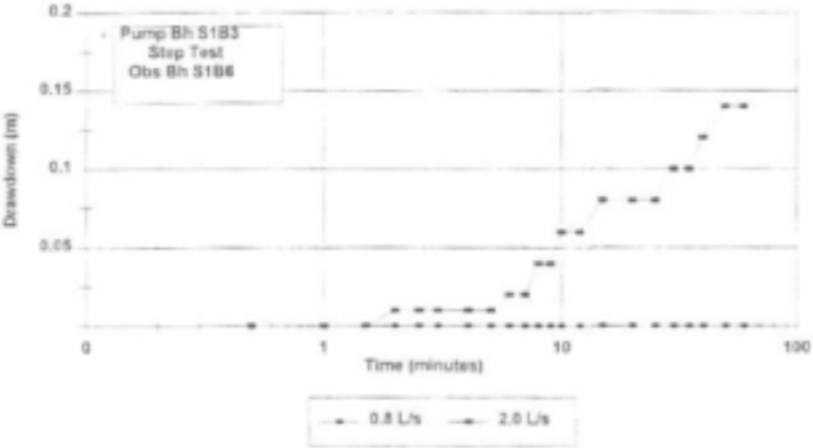


Figure 8 Observation borehole S1B6, pumping borehole S1B3 (step test)

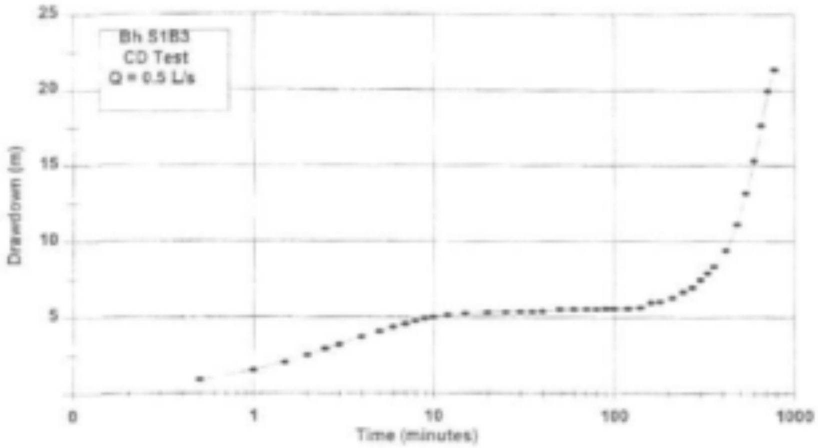


Figure 9 Borehole S1B3 constant discharge test

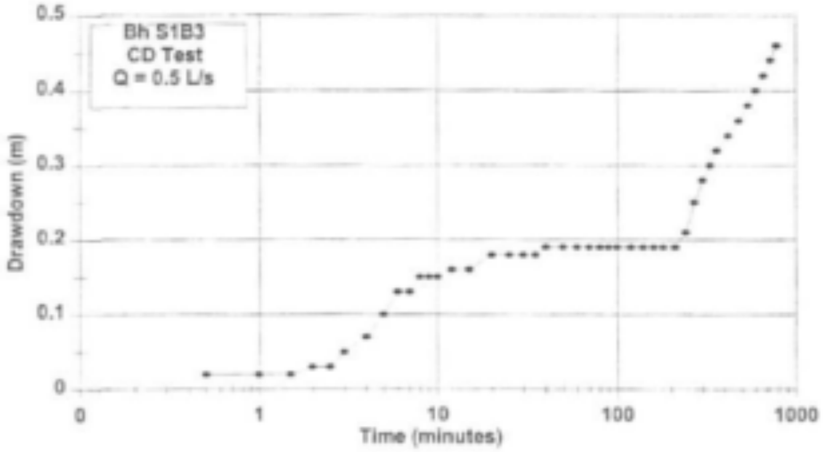


Figure 10 Observation boreholeS1B1, pumping borehole S1B3 (constant discharge test)

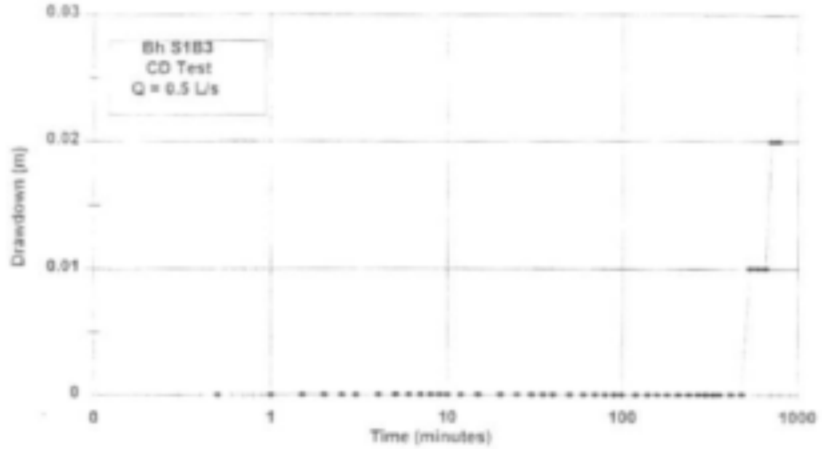


Figure 11 Observation boreholeS1B6, pumping borehole S1B3 (constant discharge test)

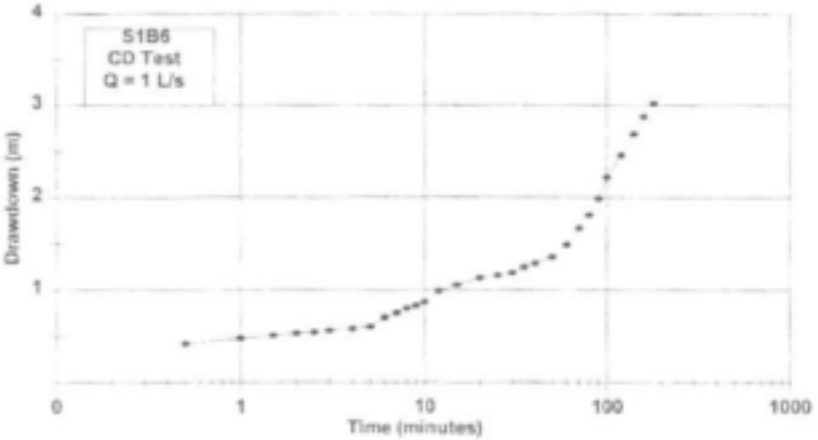


Figure 12 S1B6 constant discharge test

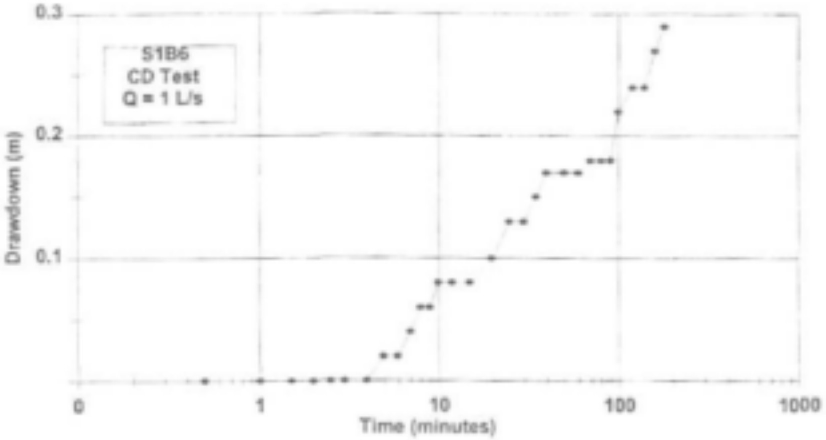
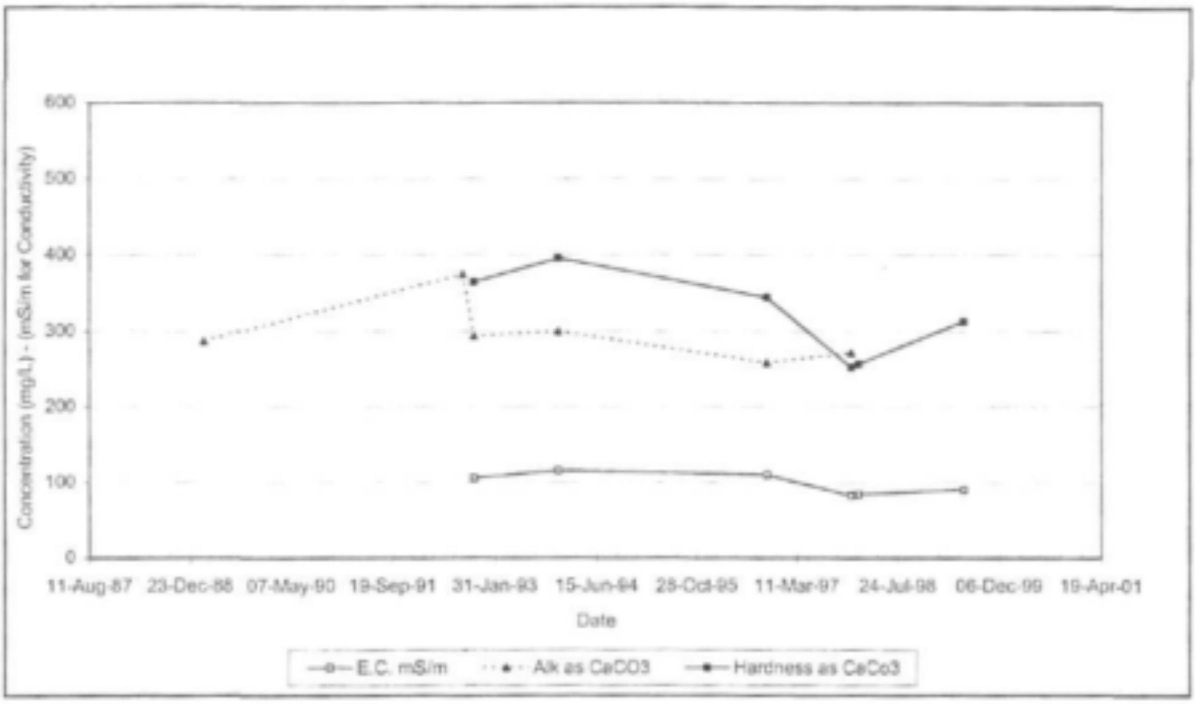


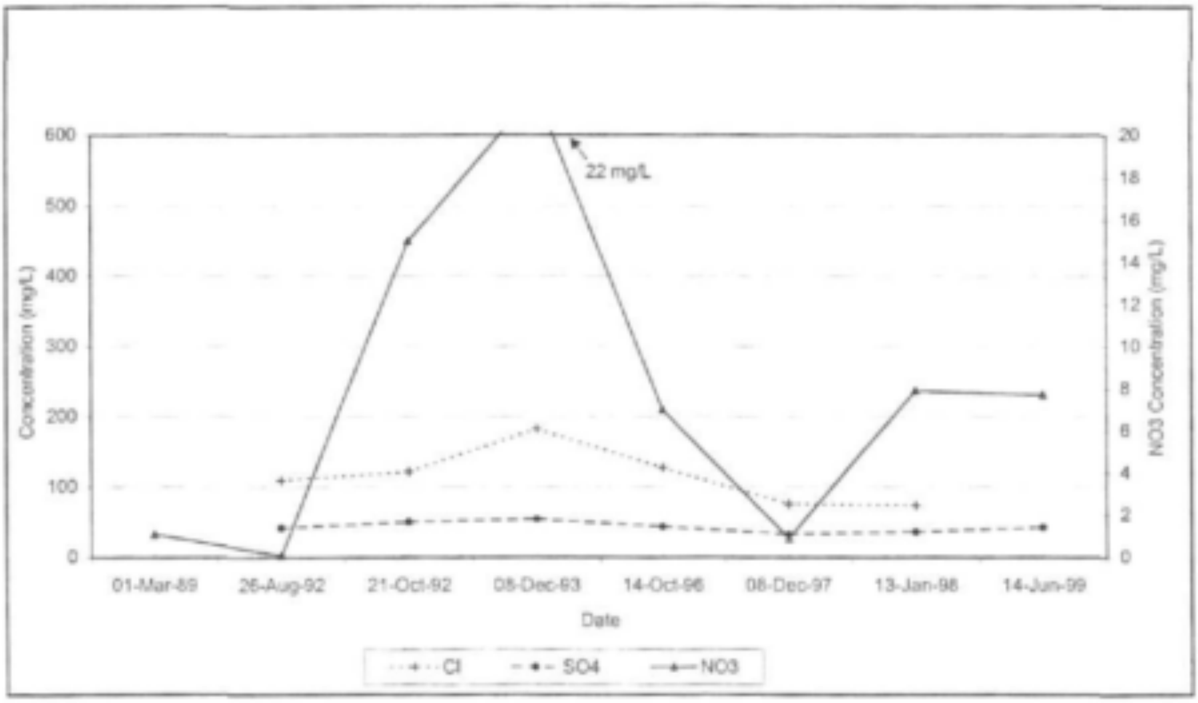
Figure 13 Observation boreholeS1B1, pumping borehole S1B6 (constant discharge test)

APPENDIX 4.3:

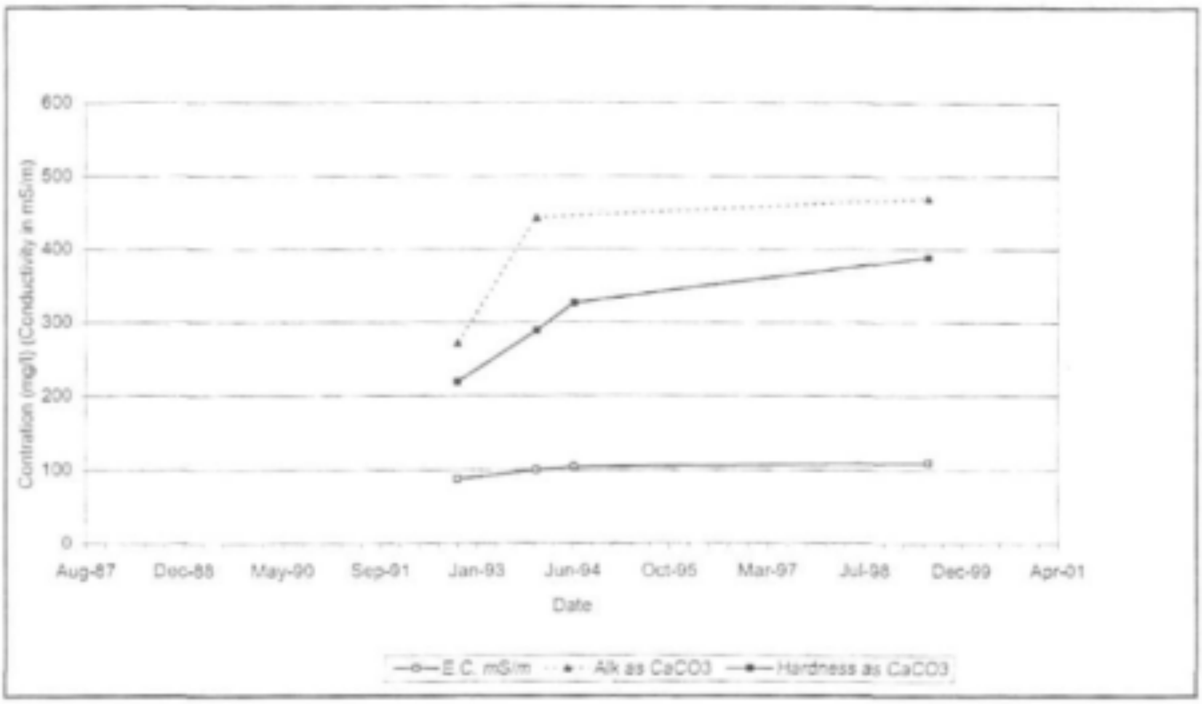
Borehole Chemistry Analysis

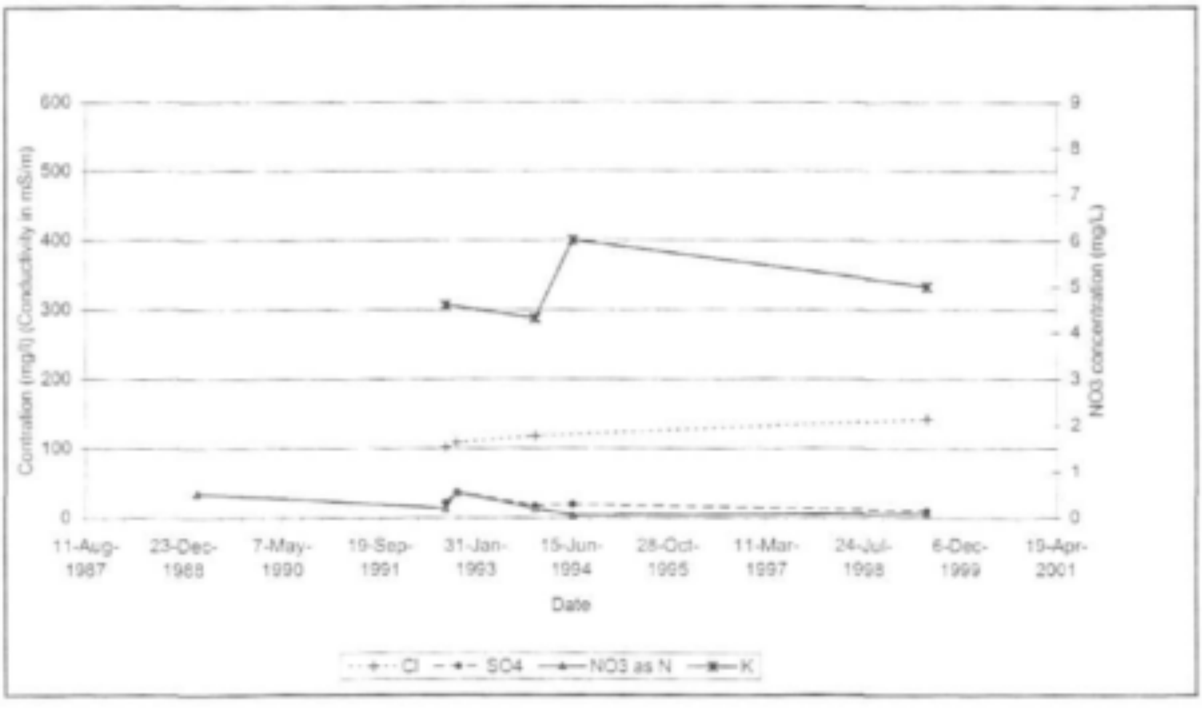
Borehole 1:


600
500
400
300
200
100
0
Concentration (mg/L) - (mS/m for Conductivity)
11-Aug-87
23-Dec-88
07-May-90
19-Sep-91
31-Jan-93
15-Jun-94
28-Oct-95
11-Mar-97
24-Jul-98
06-Dec-99
19-Apr-01
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCo3

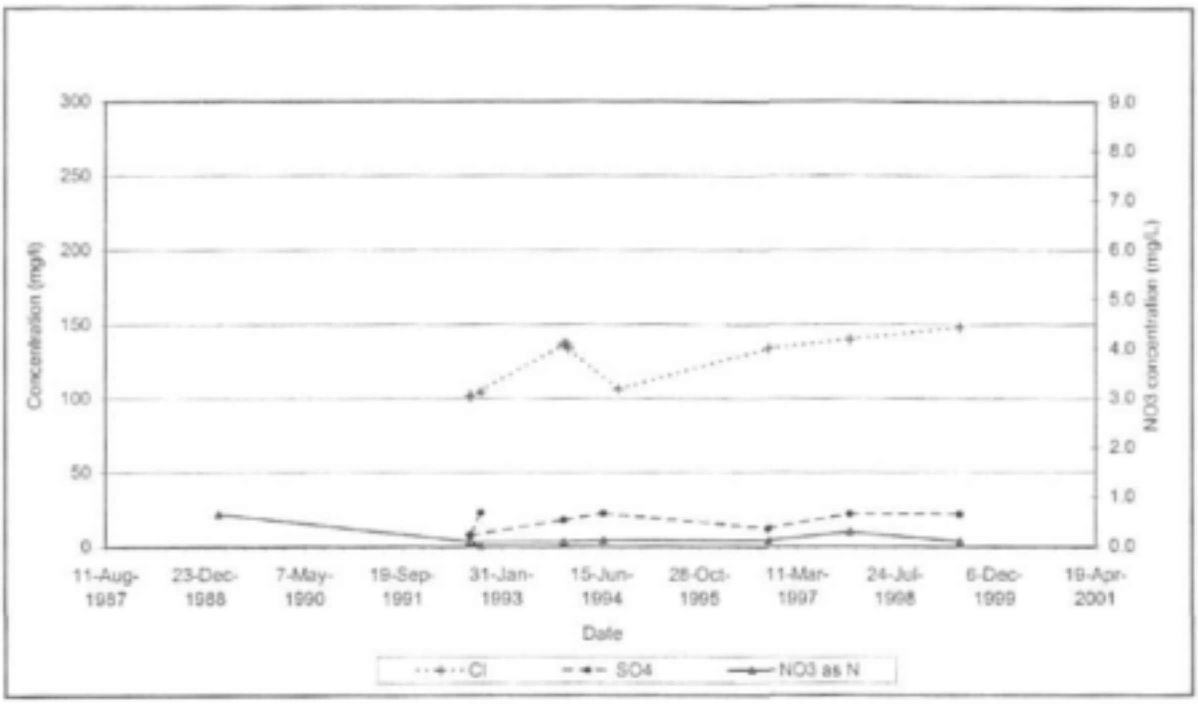

600
500
400
300
200
100
0
Concentration (mg/L)
20
18
16
14
12
10
8
6
4
2
0
NO3 Concentration (mg/L)
22 mg/L
01-Mar-89
26-Aug-92
21-Oct-92
08-Dec-93
14-Oct-96
08-Dec-97
13-Jan-98
14-Jun-99
Date
Cl
SO4
NO3

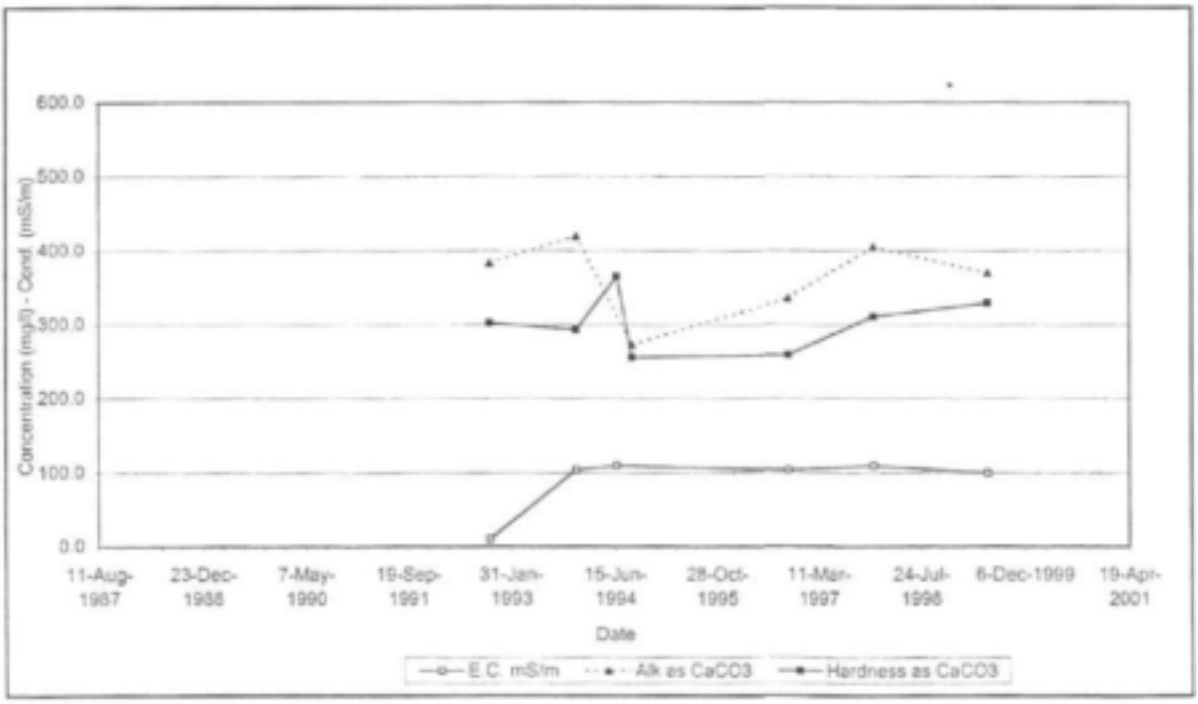
Borehole 2:


600
500
400
300
200
100
0
Contration (mg/l) (Conductivity in mS/m)
Aug-87
Dec-88
May-90
Sep-91
Jan-93
Jun-94
Oct-95
Mar-97
Jul-98
Dec-99
Apr-01
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

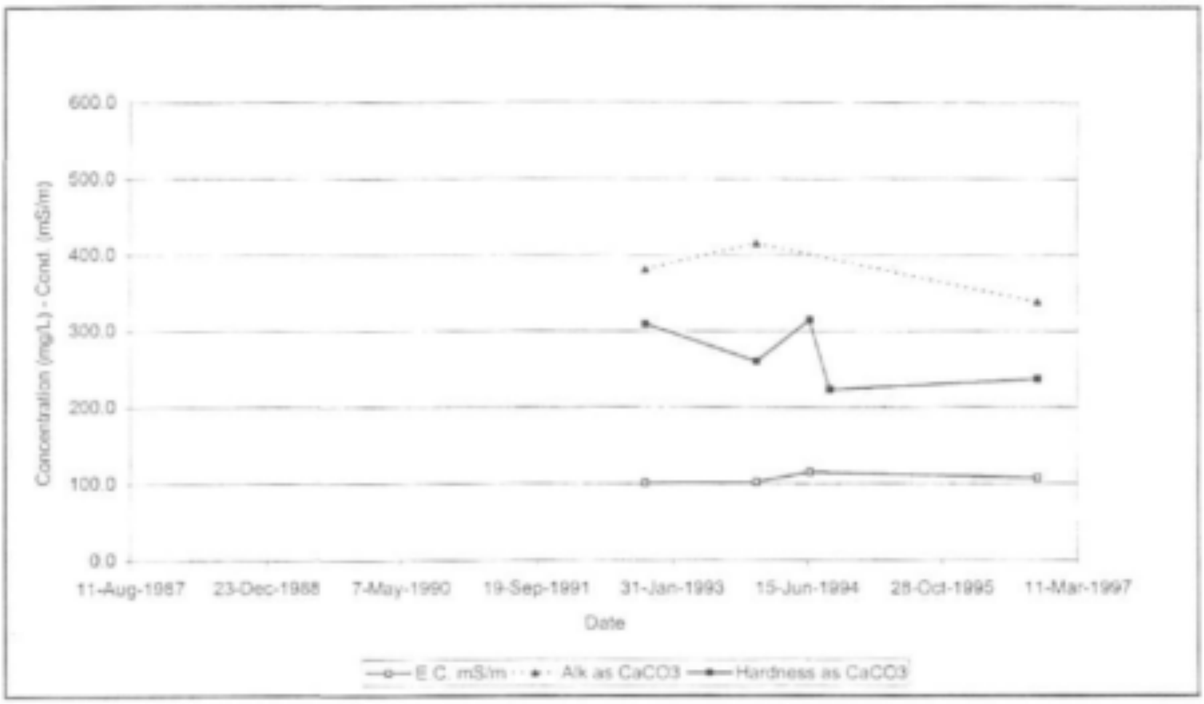

600
500
400
300
200
100
0
Contration (mg/l) (Conductivity in mS/m)
9
8
7
6
5
4
3
2
1
0
NO3 concentration (mg/L)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
19-Apr-2001
Date
Cl
SO4
NO3 as N
K

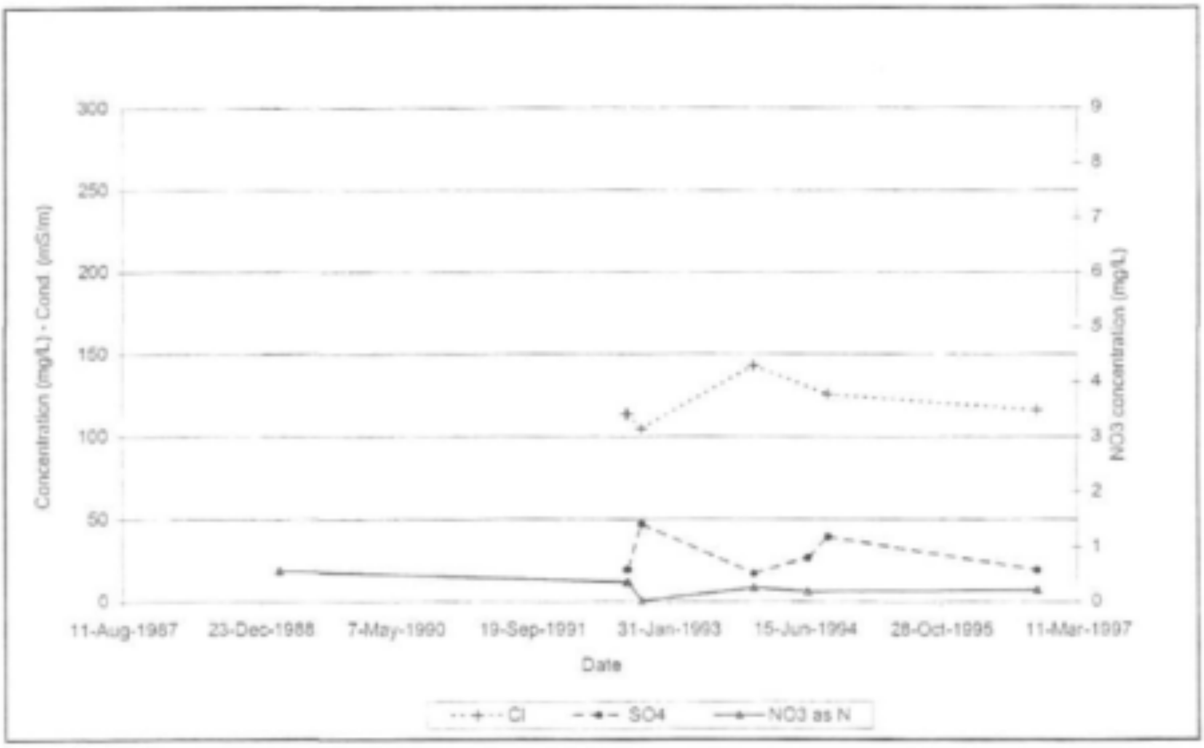
Borehole 3:


300
250
200
150
100
50
0
Concentration (mg/l)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
NO3 concentration (mg/L)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
19-Apr-2001
Date
Cl
SO4
NO3 as N

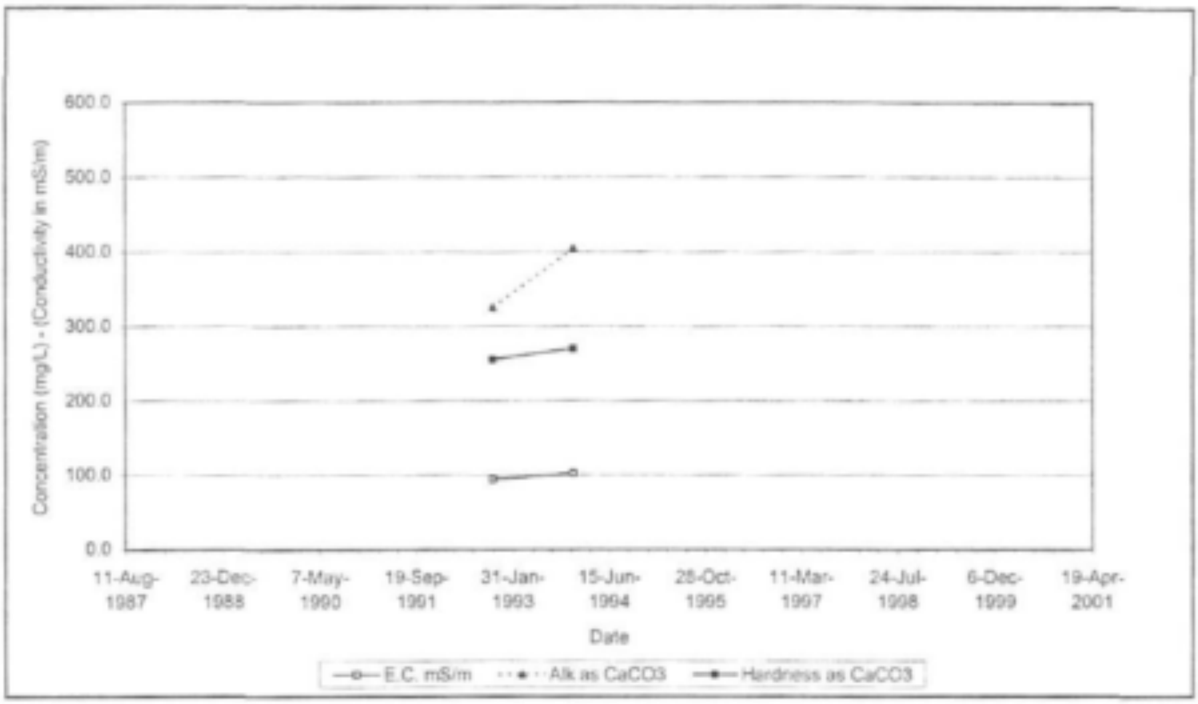

600.0
500.0
400.0
300.0
200.0
100.0
0.0
Concentration (mg/l) - Cond. (mS/m)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
19-Apr-2001
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

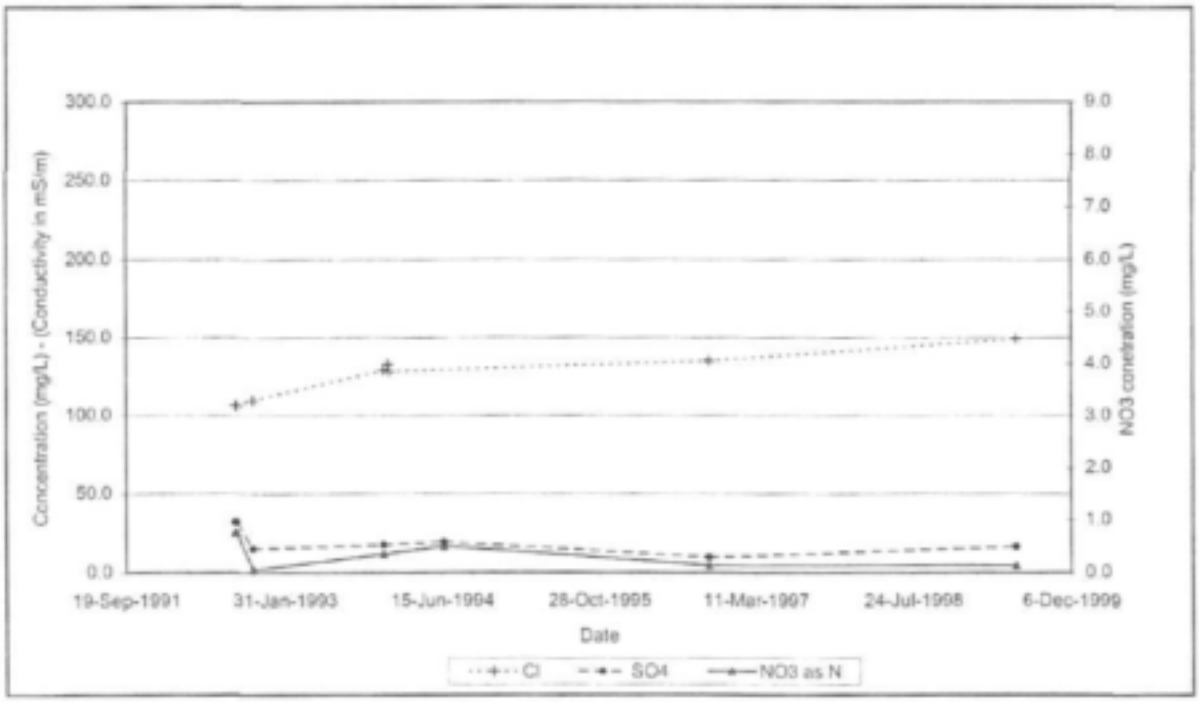
Borehole 4:


600.0
500.0
400.0
300.0
200.0
100.0
0.0
Concentration (mg/L) - Cond. (mS/m)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

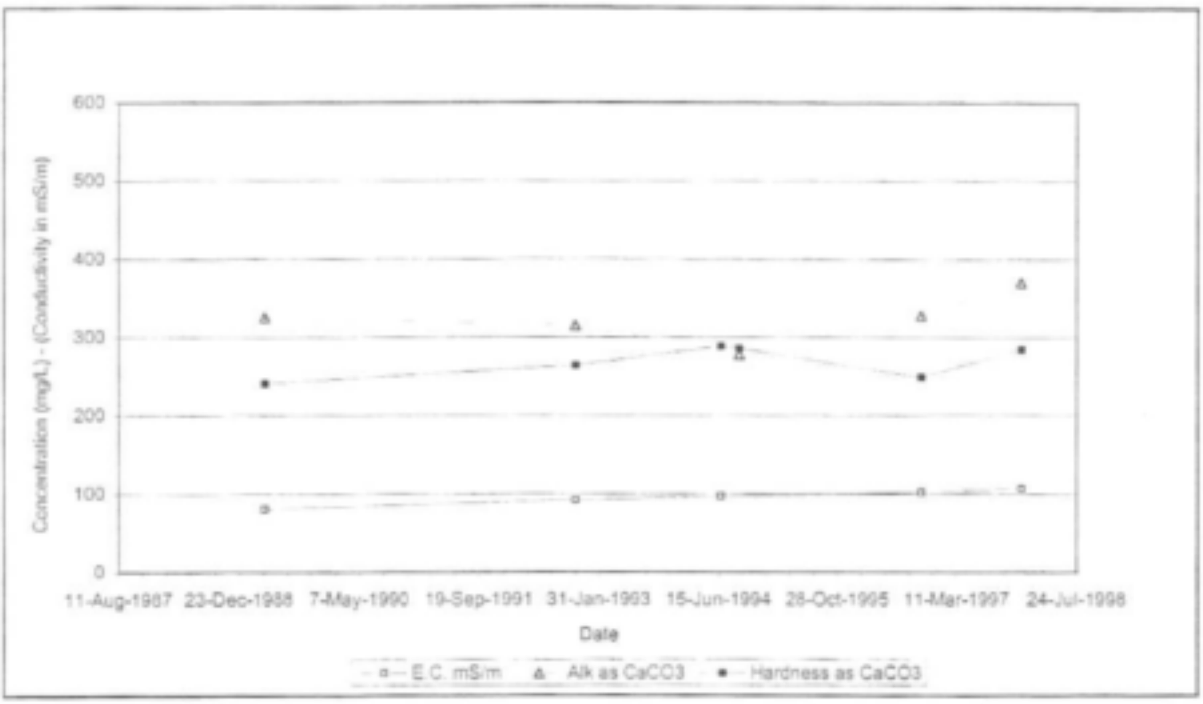

300
250
200
150
100
50
0
Concentration (mg/L) - Cond. (mS/m)
9
8
7
6
5
4
3
2
1
0
NO3 concentration (mg/L)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
Date
Cl
SO4
NO3 as N

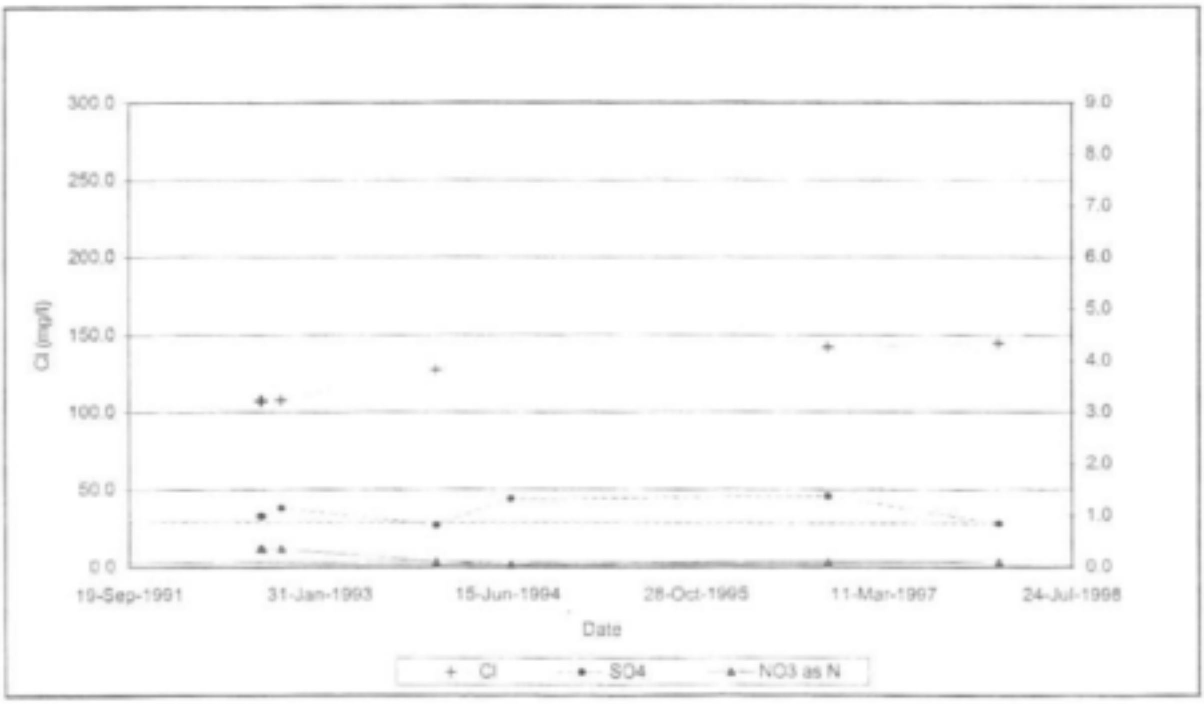
Borehole 5:


Concentration (mg/L) - (Conductivity in mS/m)
600.0
500.0
400.0
300.0
200.0
100.0
0.0
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
19-Apr-2001
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

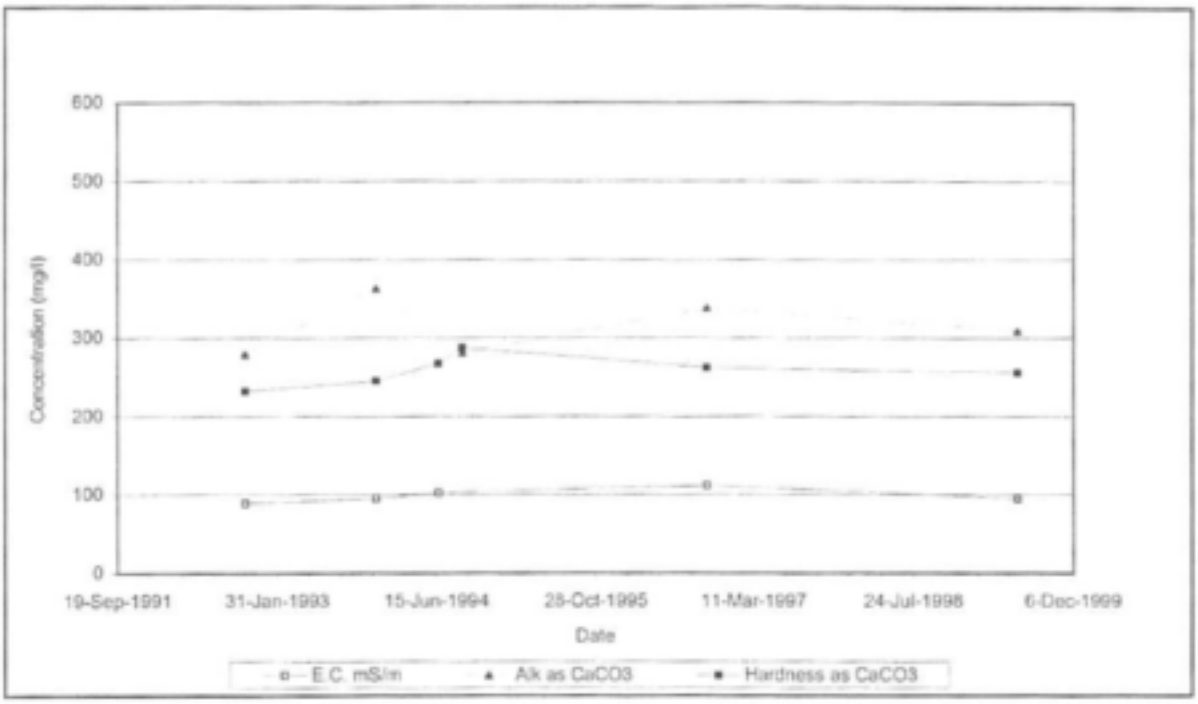

Concentration (mg/L) - (Conductivity in mS/m)
300.0
250.0
200.0
150.0
100.0
50.0
0.0
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
NO3 conetration (mg/L)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
Cl
SO4
NO3 as N

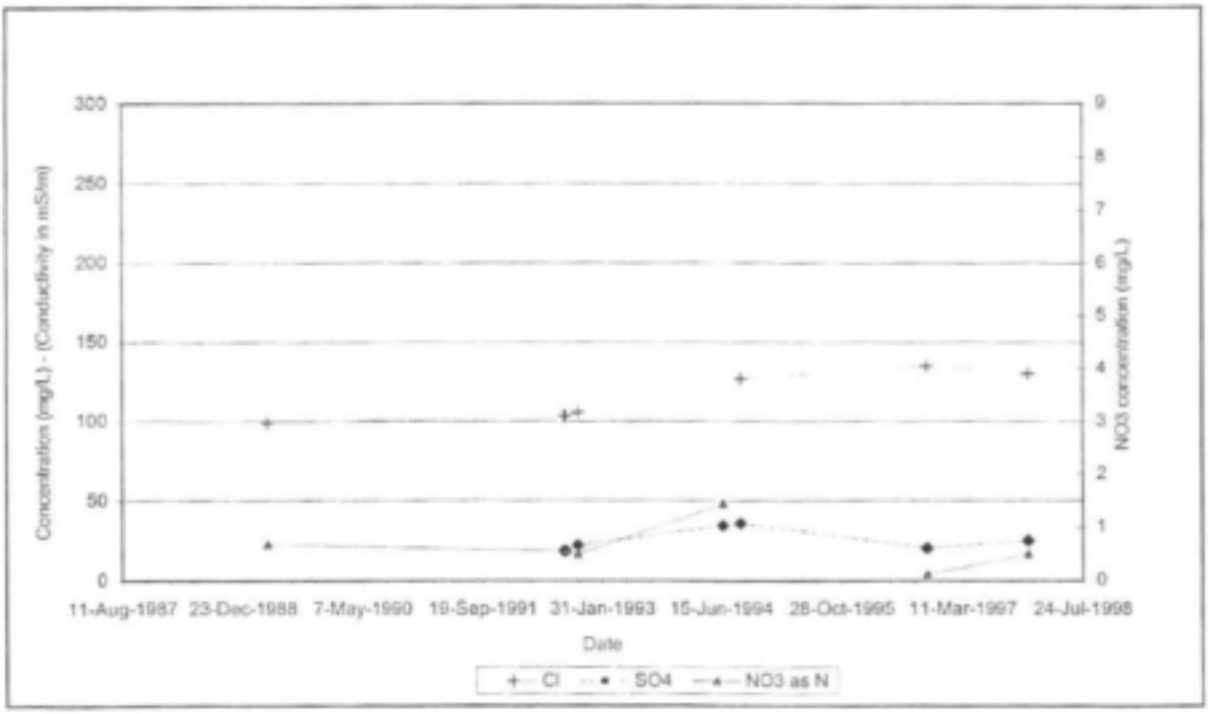
Borehole 6:


600
500
400
300
200
100
0
Concentration (mg/L) - (Conductivity in mS/m)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

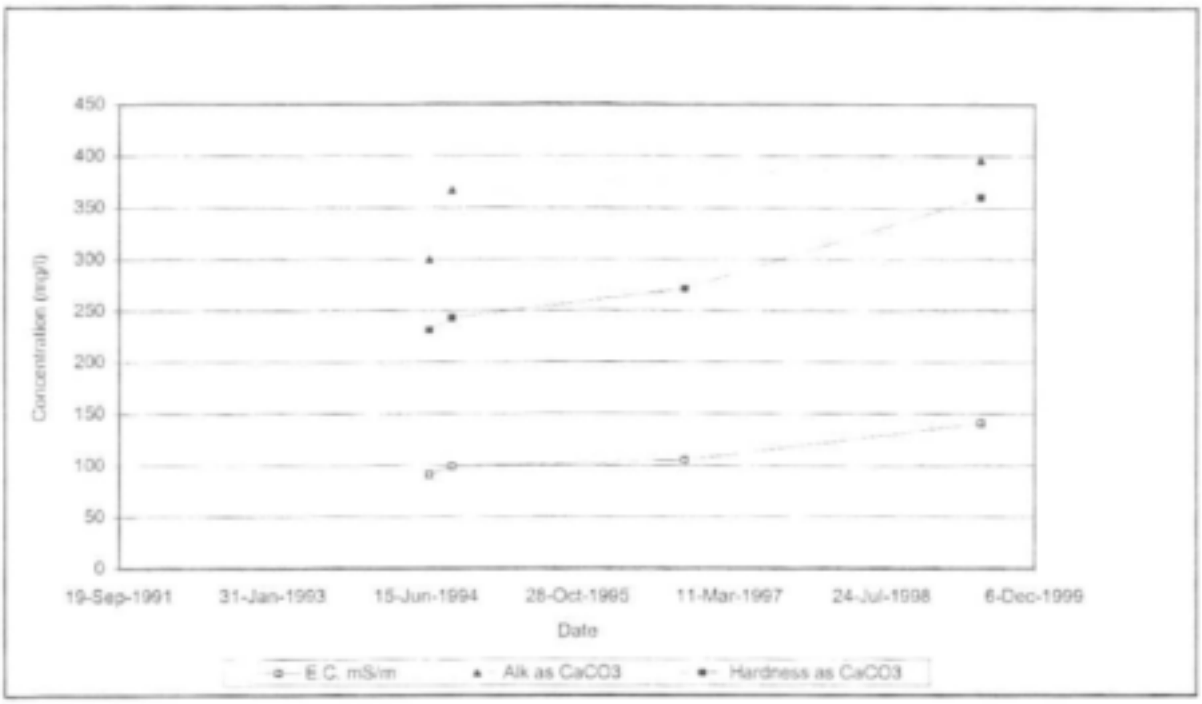

300.0
250.0
200.0
150.0
100.0
50.0
0.0
Cl (mg/l)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
Date
Cl
SO4
NO3 as N

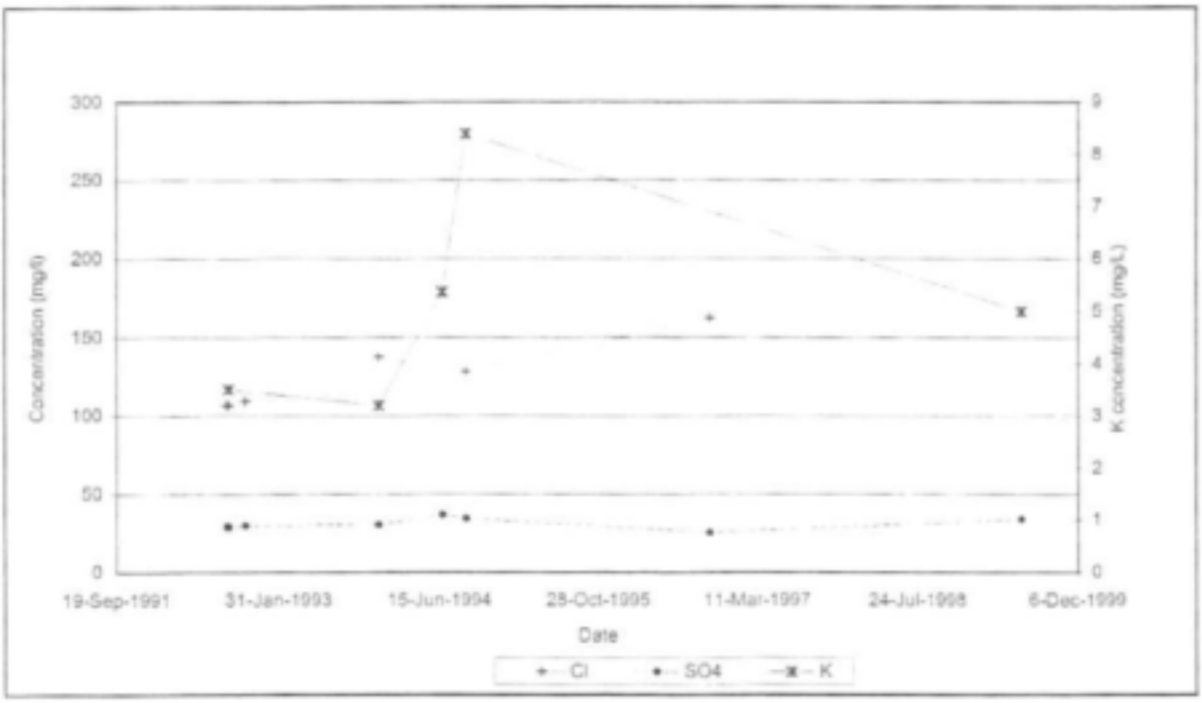
Borehole 7:


600
500
400
300
200
100
0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

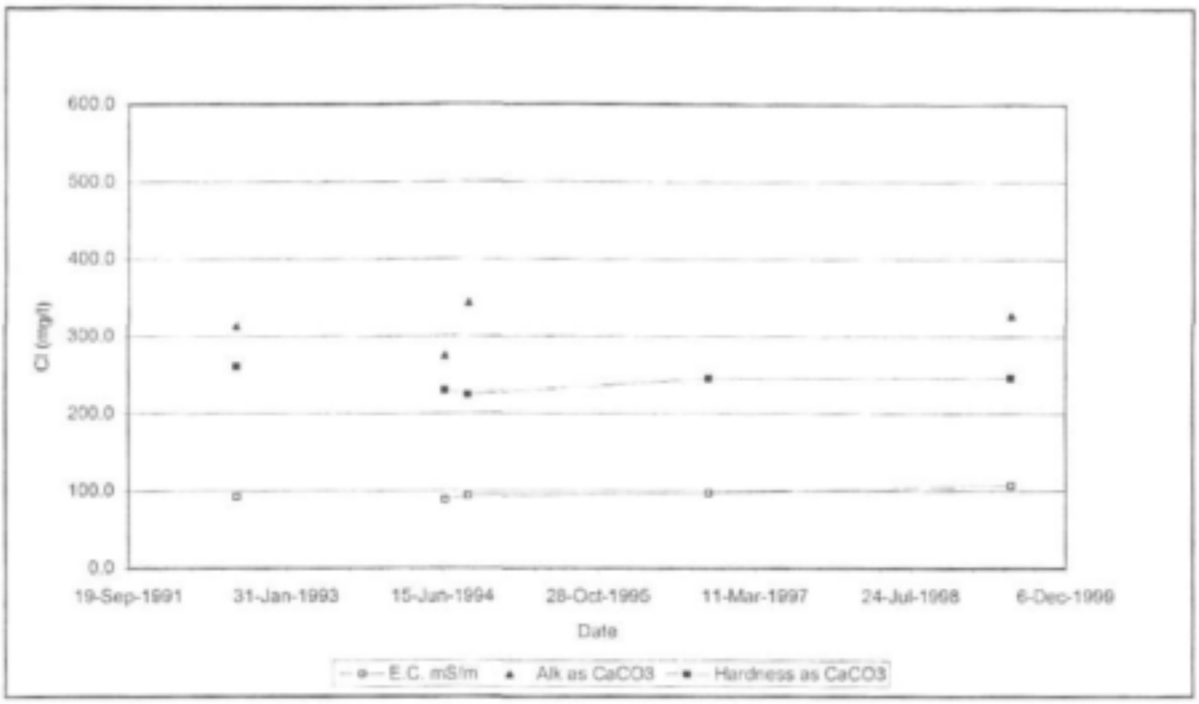

300
250
200
150
100
50
0
Concentration (mg/L) - (Conductivity in mS/m)
9
8
7
6
5
4
3
2
1
0
NO3 concentration (mg/L)
11-Aug-1987
23-Dec-1988
7-May-1990
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
Date
Cl
SO4
NO3 as N

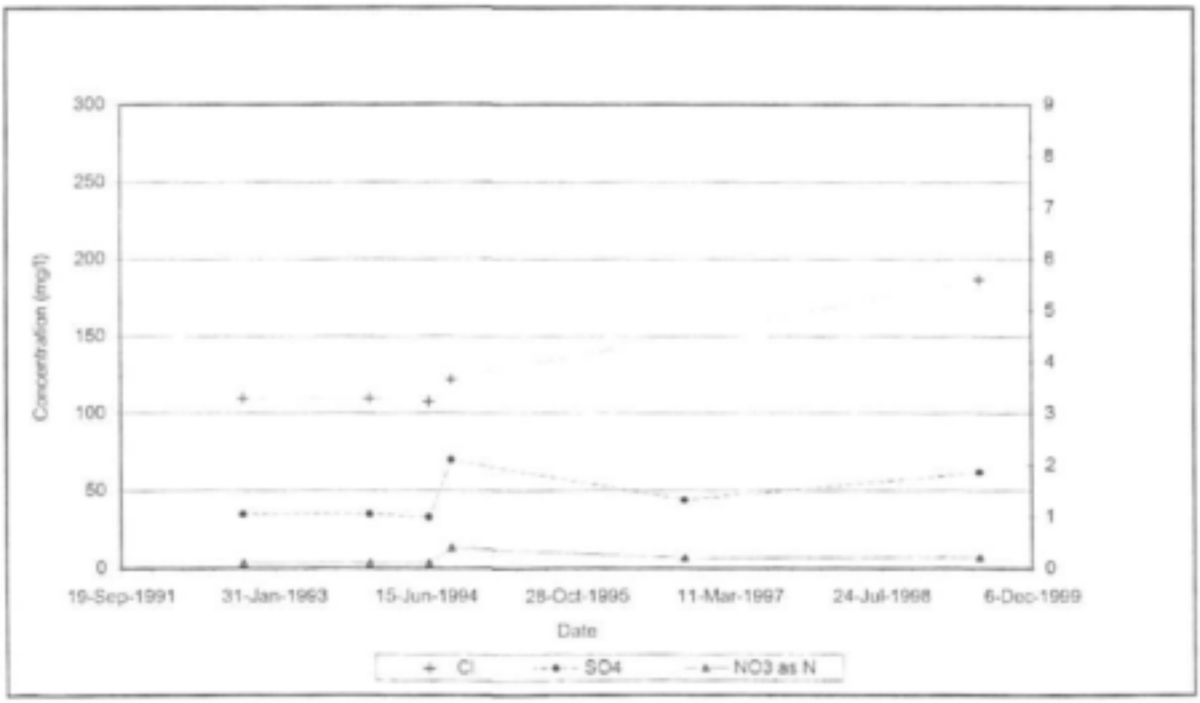
Borehole 8:


450
400
350
300
250
200
150
100
50
0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

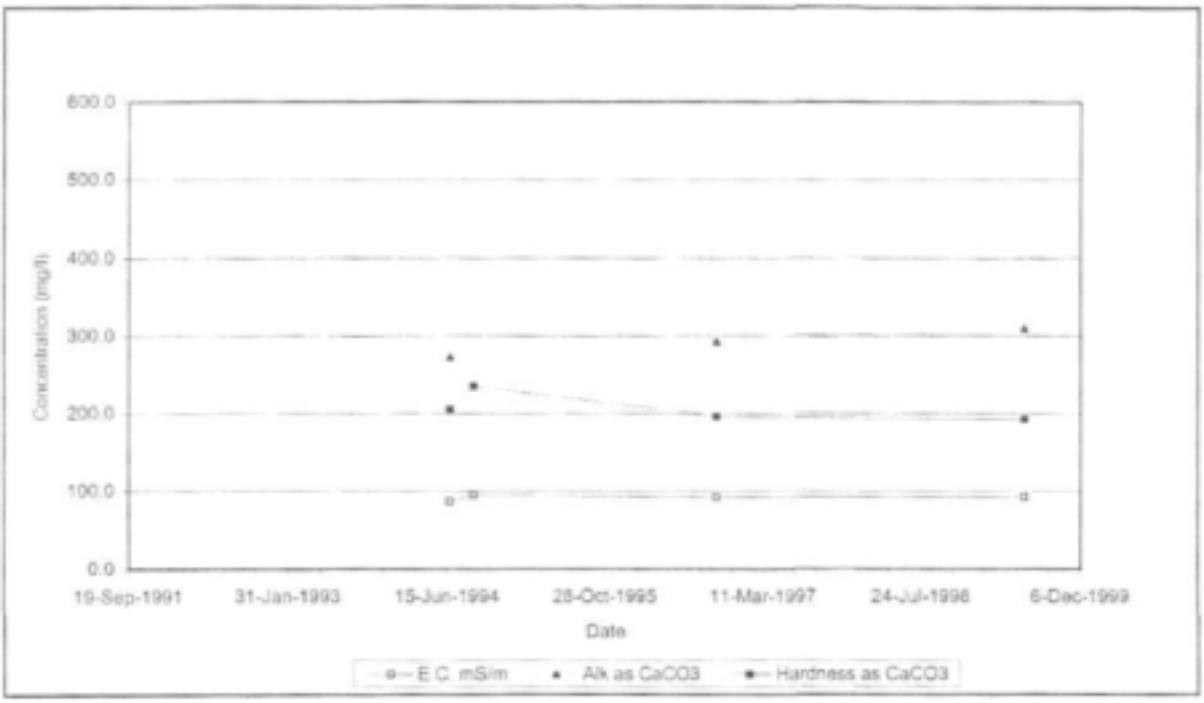

300
250
200
150
100
50
0
Concentration (mg/l)
9
8
7
6
5
4
3
2
1
0
K concentration (mg/L)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
Cl
SO4
K

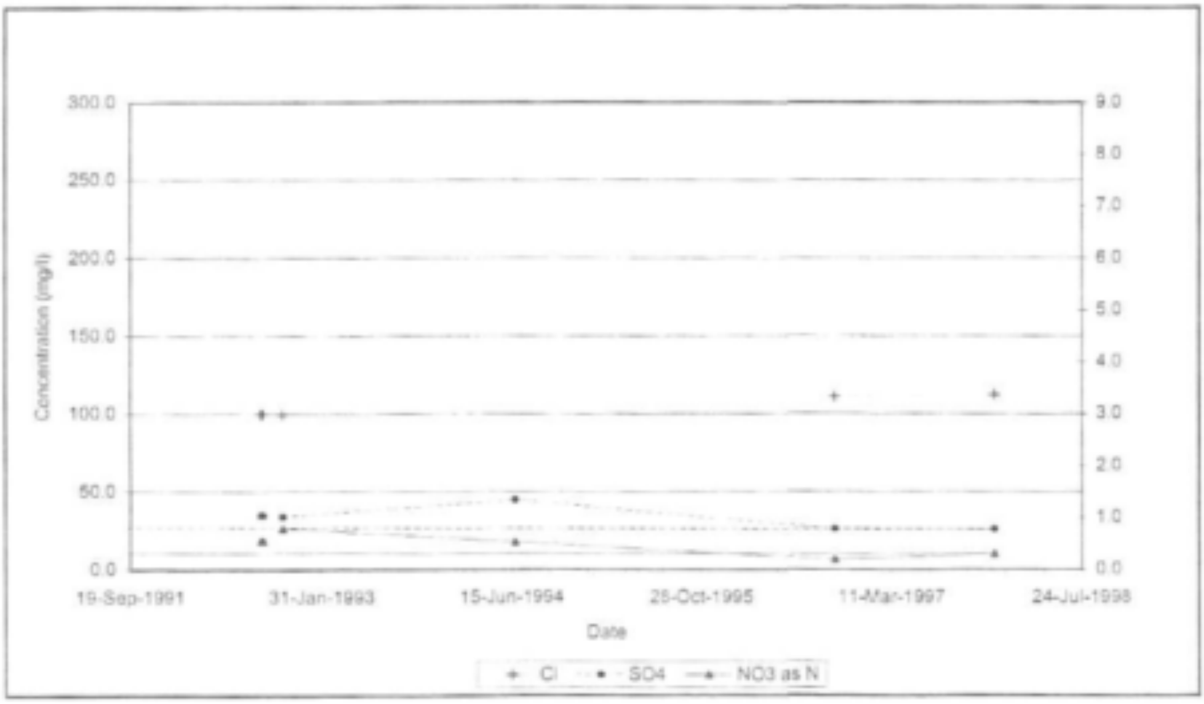
Borehole 9:


600.0
500.0
400.0
300.0
200.0
100.0
0.0
Cl (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3


300
250
200
150
100
50
0
9
8
7
6
5
4
3
2
1
0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
Cl
SO4
NO3 as N

Borehole 10:


600.0
500.0
400.0
300.0
200.0
100.0
0.0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3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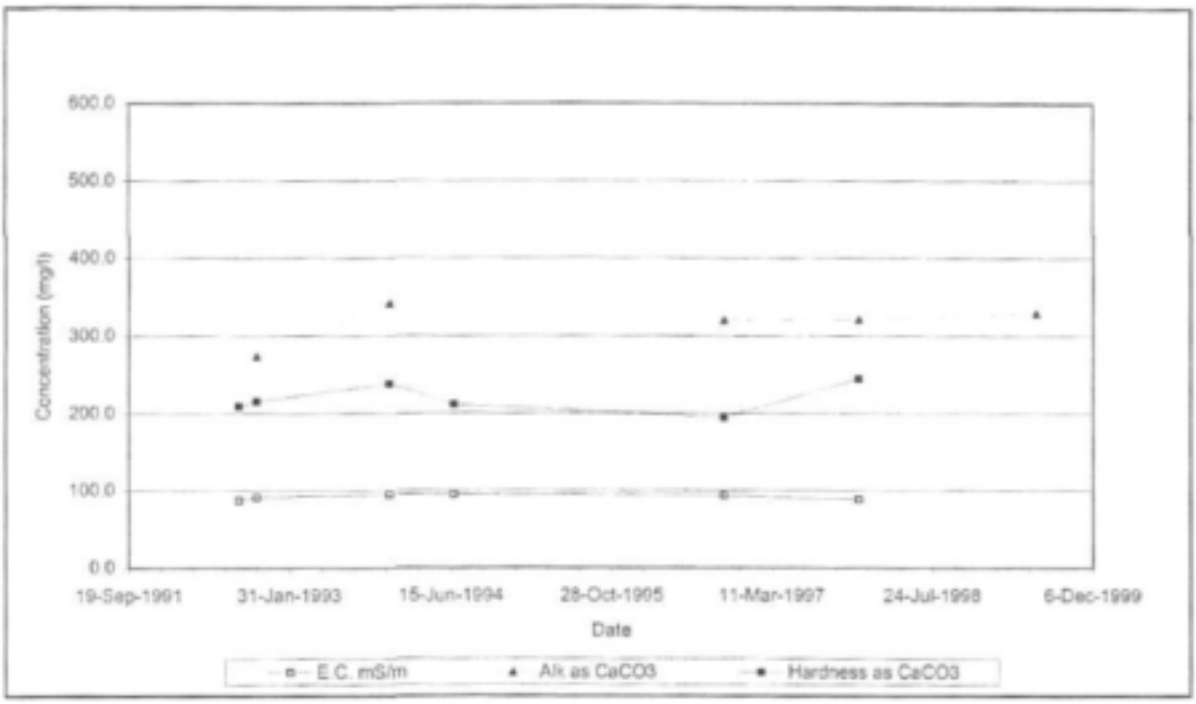
250.0
200.0
150.0
100.0
50.0
0.0
Concentration (mg/l)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
19-Sep-1991
31-Jan-1993
15-Jun-1994
25-Oct-1995
11-Mar-1997
24-Jul-1998
Date
Cl
SO4
NO3 as N

Borehole 11: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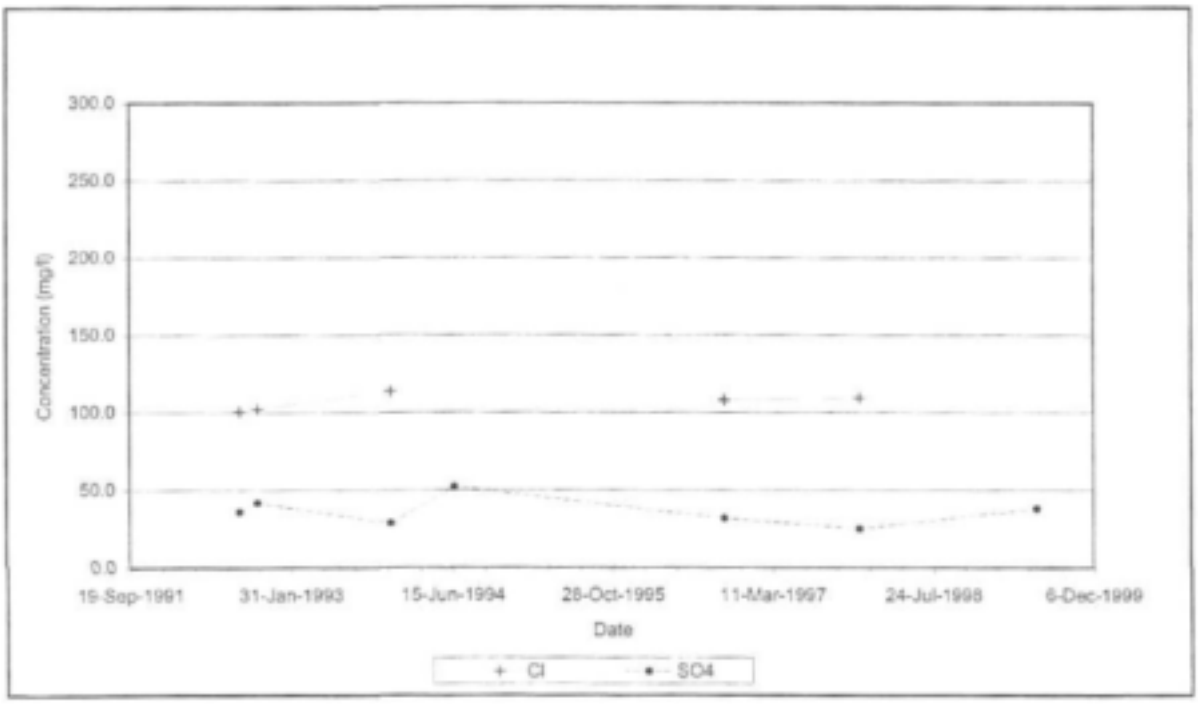
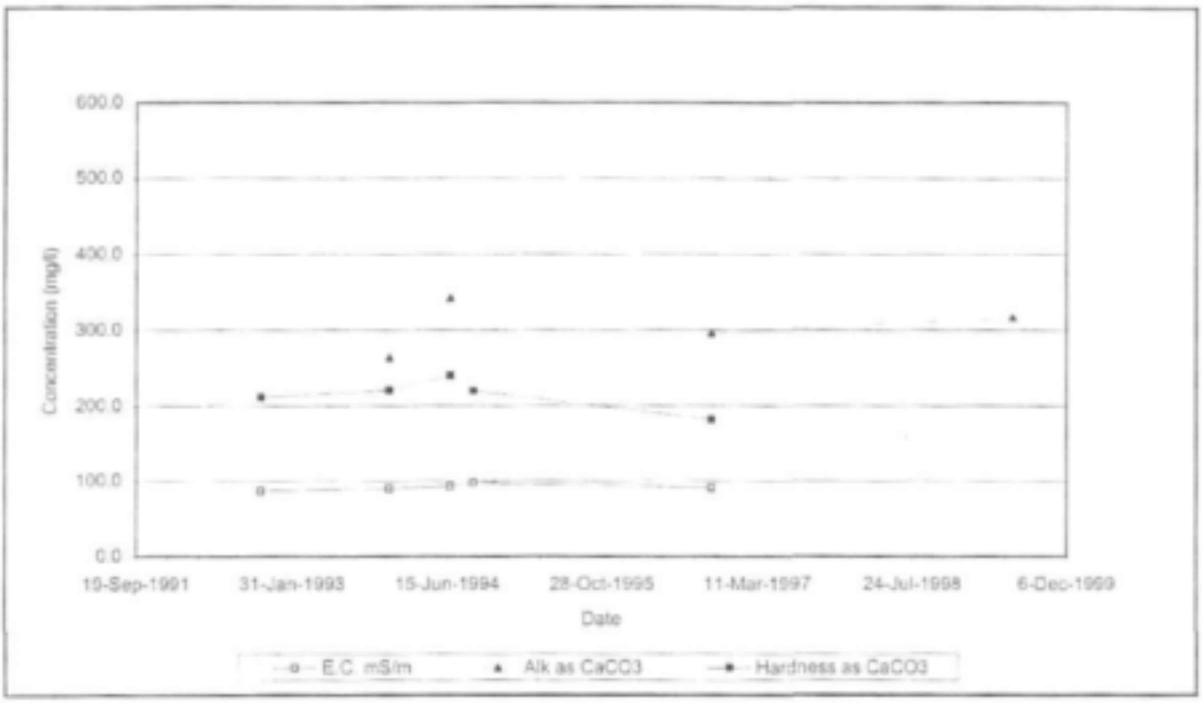
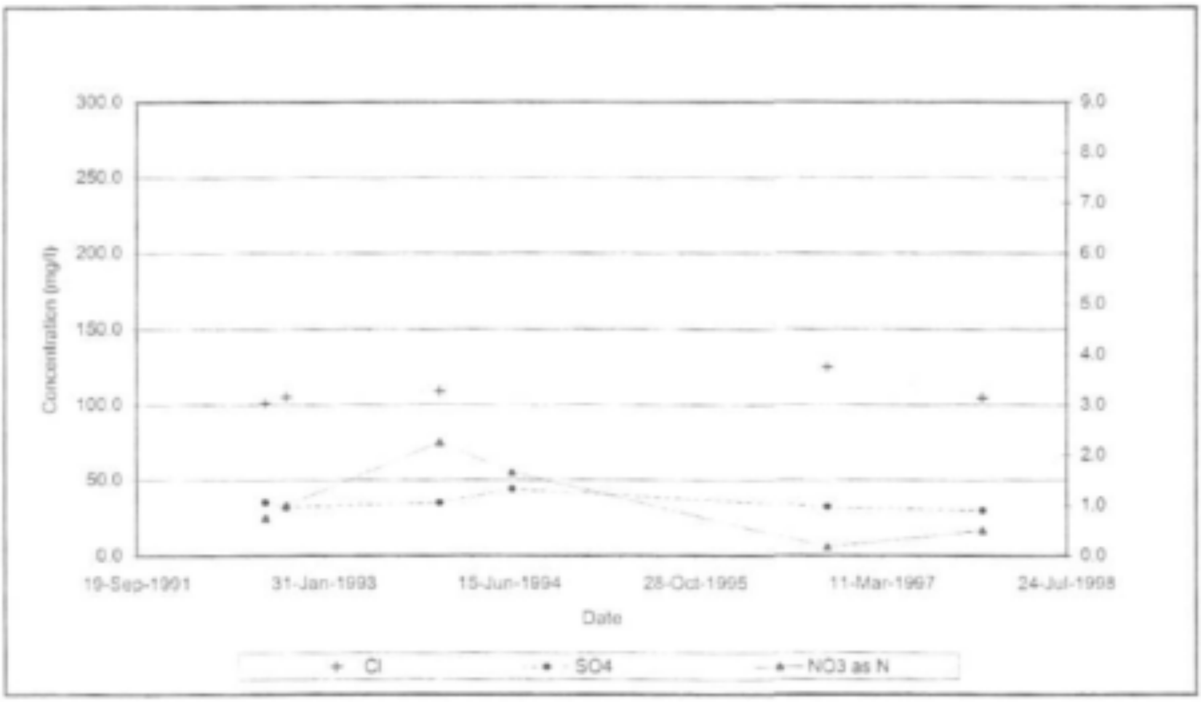
500.0
400.0
300.0
200.0
100.0
0.0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

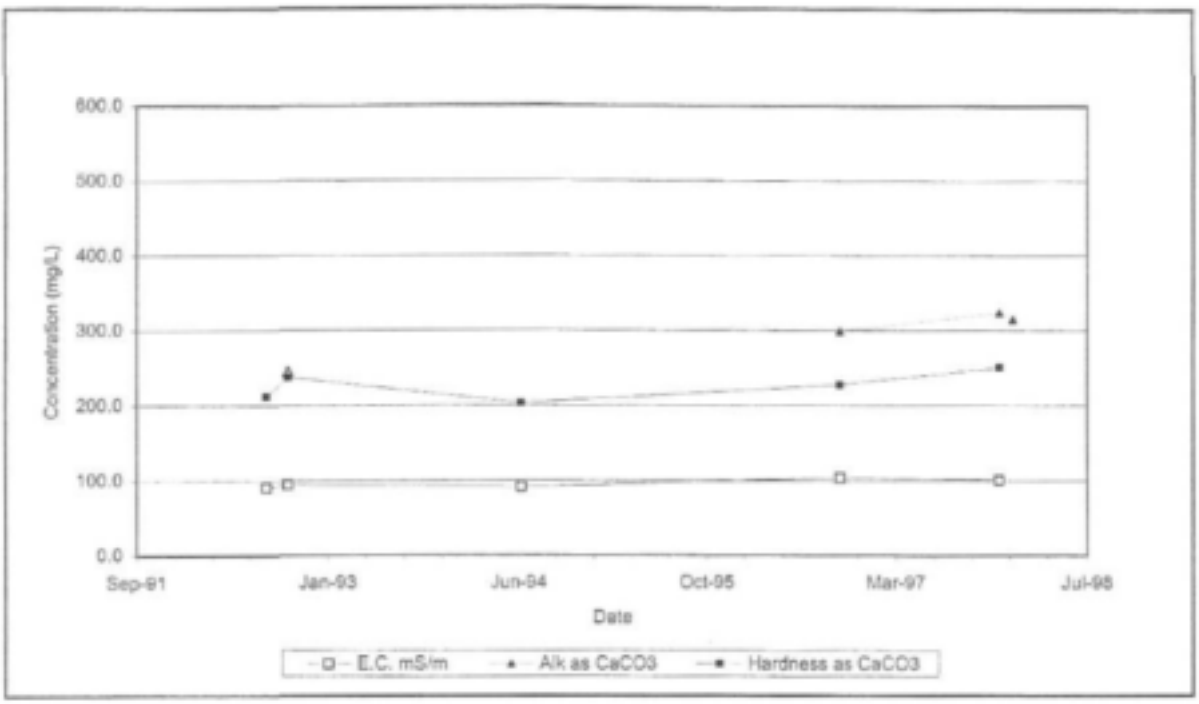

300.0
250.0
200.0
150.0
100.0
50.0
0.0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
Cl
SO4

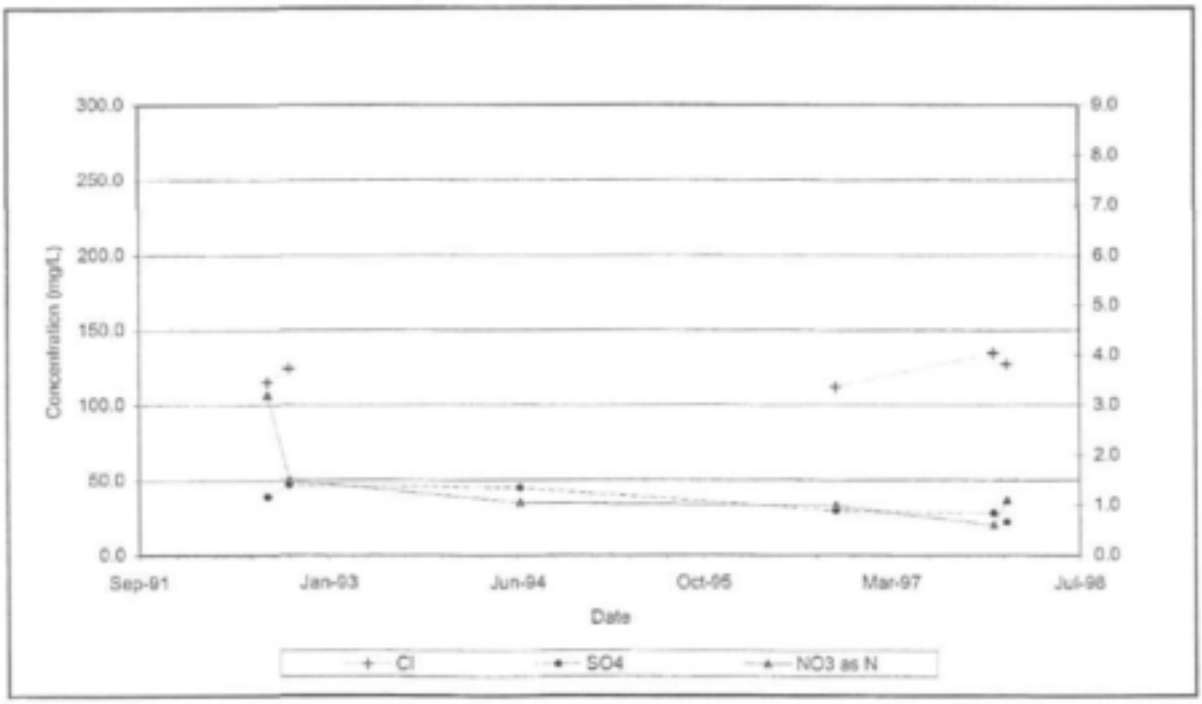
Borehole 12:


600.0
500.0
400.0
300.0
200.0
100.0
0.0
Concentration (mg/l)
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
6-Dec-1999
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3


300.0
250.0
200.0
150.0
100.0
50.0
0.0
Concentration (mg/l)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
19-Sep-1991
31-Jan-1993
15-Jun-1994
28-Oct-1995
11-Mar-1997
24-Jul-1998
Date
Cl
SO4
NO3 as N

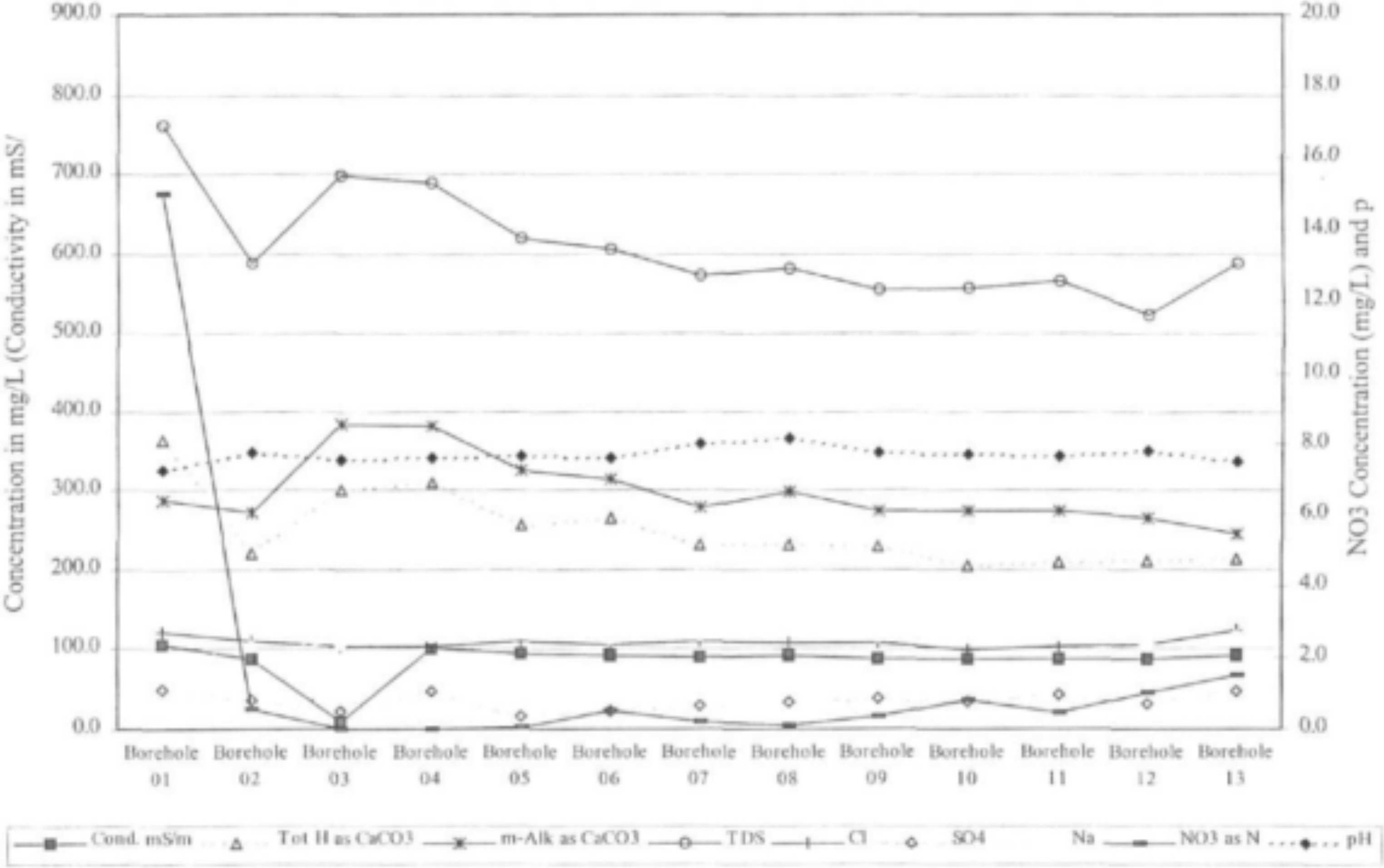
Borehole 13:


600.0
500.0
400.0
300.0
200.0
100.0
0.0
Concentration (mg/L)
Sep-91
Jan-93
Jun-94
Oct-95
Mar-97
Jul-98
Date
E.C. mS/m
Alk as CaCO3
Hardness as CaCO3

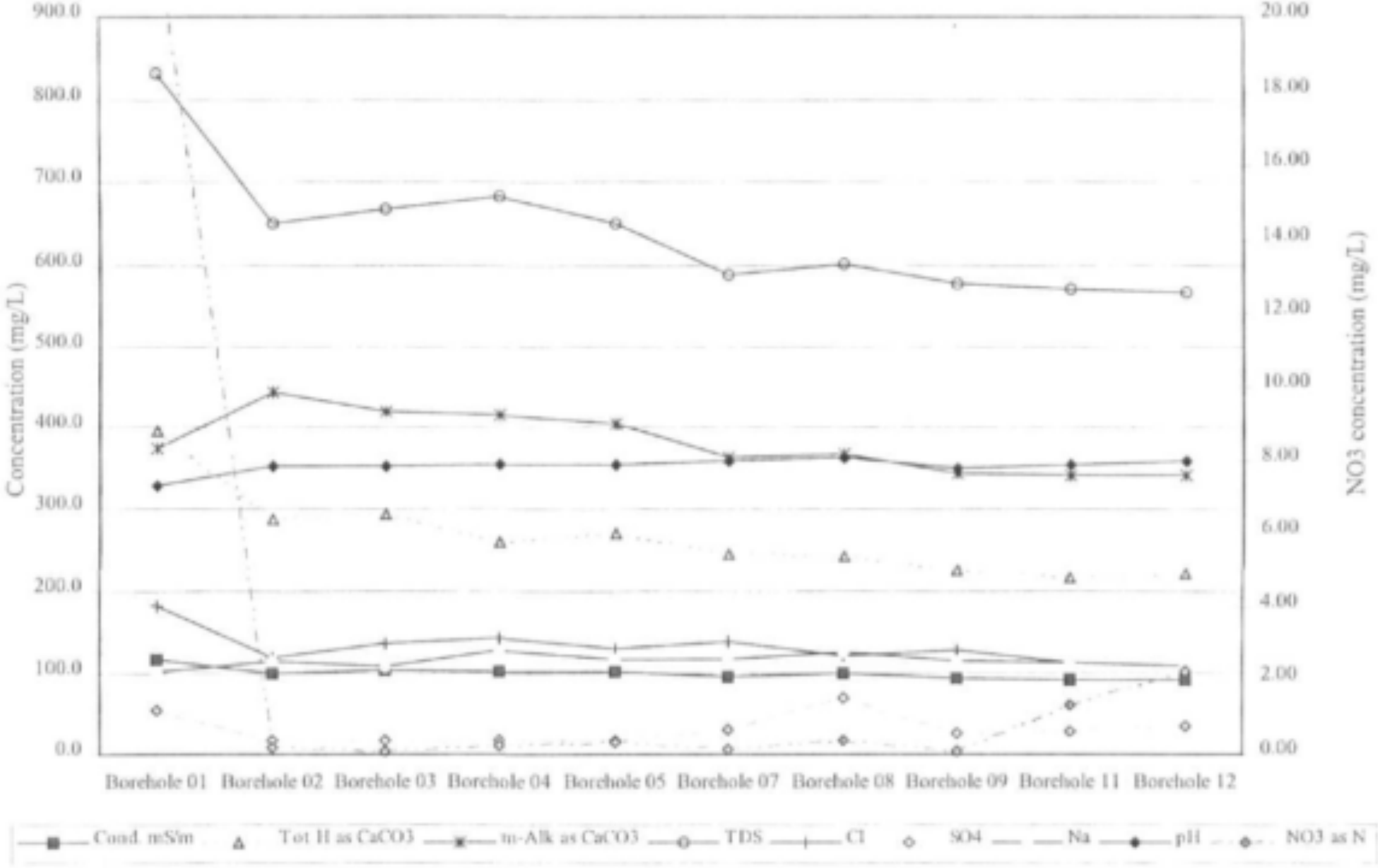

300.0
250.0
200.0
150.0
100.0
50.0
0.0
Concentration (mg/L)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
Sep-91
Jan-93
Jun-94
Oct-95
Mar-97
Jul-98
Date
Cl
SO4
NO3 as N

APPENDIX 4.4:

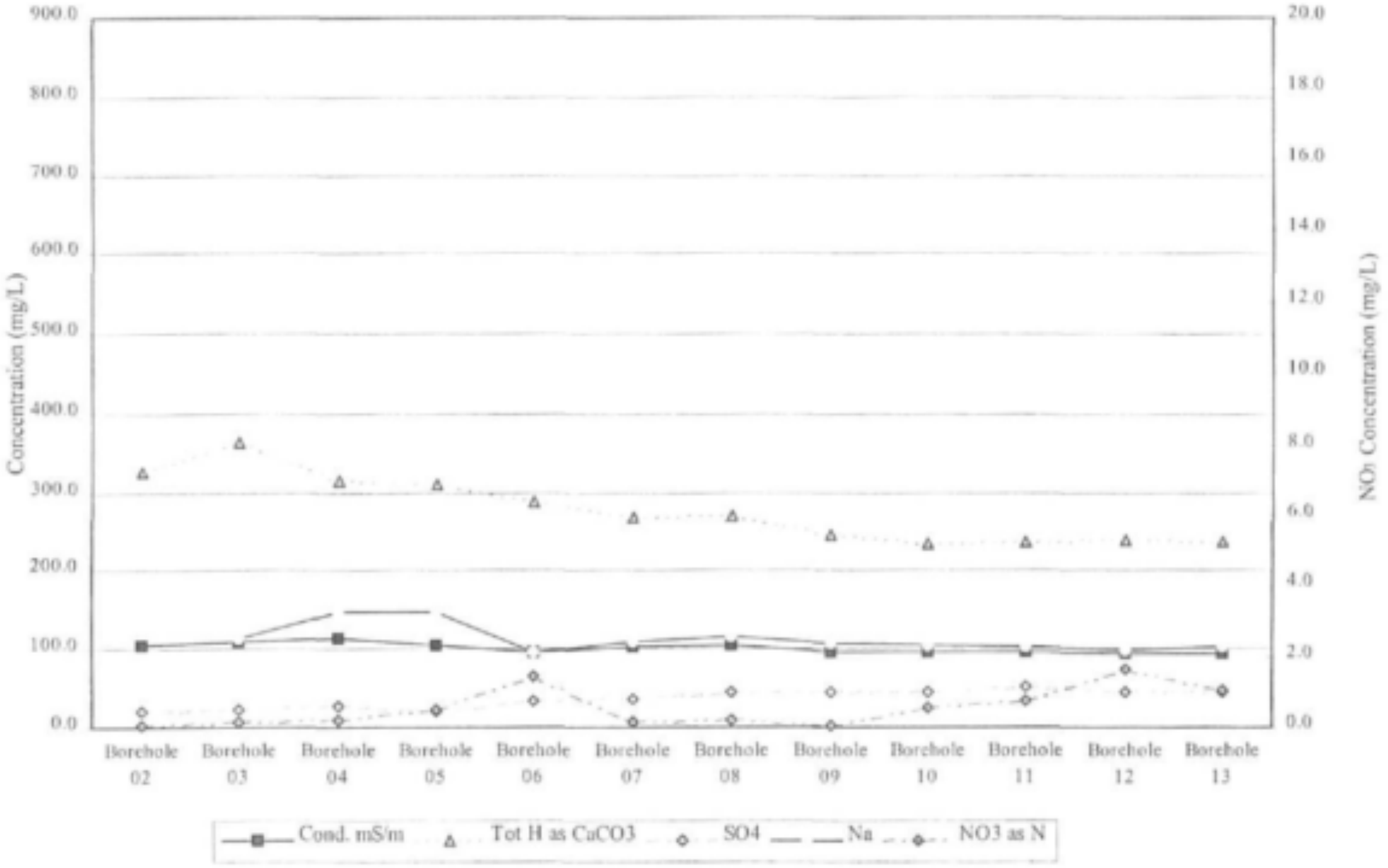
Borehole Chemistry with Relative Distance from WWTW


Change in Water quality with distance from WWTW (21/10/92)

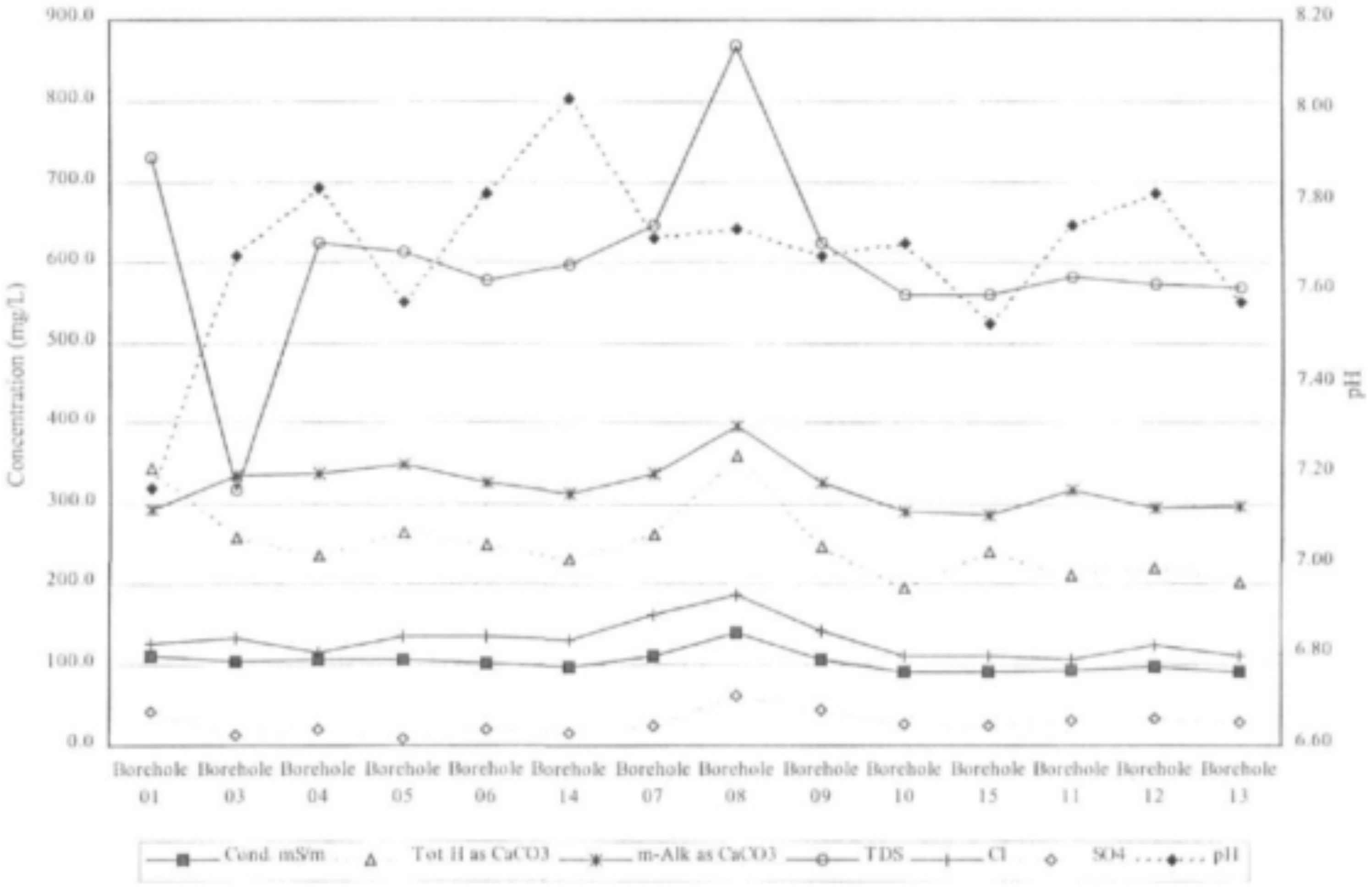
Change in Water quality with distance from WWTW (08/12/93)



Change in Water Quality with Distance from WWTW (23/06/94)



Change in Water Quality with distance from WWTW (14/10/96)



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